

# **NOTICE**

**All drawings located at the end of the document.**

**DRAFT FINAL**

**Phase III**  
**RFI/RI Report**

**Rocky Flats Plant**  
**881 Hillside Area**  
(Operable Unit No. 1)

***Volume XIII***  
***Appendix E***  
***Environmental Evaluation***

**U.S. Department of Energy**  
**Rocky Flats Plant**  
**Golden, Colorado**

REVIEWED FOR CLASSIFICATION/UCN

By

*[Signature]*

October 1992

ADMIN RECORD

Date

*11/4/92* *[Signature]*

A-DU01-000451

## Appendix E

Environmental  
Evaluation

Phase III  
RFI/RI Report

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By [Signature]  
Date 11/4/92 [Signature]

PHASE III RFI/RI REPORT

APPENDIX E  
ENVIRONMENTAL EVALUATION

Rocky Flats Plant  
881 Hillside Area

Operable Unit No. 1

U.S. Department of Energy  
Rocky Flats Plant  
Golden, Colorado

October 1992



## TABLE OF CONTENTS

<u>Section</u>	<u>Page</u>
<u>EXECUTIVE SUMMARY</u> .....	xv
<u>E1.0 INTRODUCTION</u> .....	E-1
E1.1 PURPOSE .....	E-1
E1.2 FOCUS OF OU1 ENVIRONMENTAL EVALUATIONS .....	E-1
E1.2.1 <u>Guidelines Used for Structuring Environmental Evaluation</u> .....	E-1
E1.2.2 <u>Definition and Concept of Ecological Risk Assessment</u> .....	E-2
E1.3 OBJECTIVES .....	E-2
E1.4 SCOPE OF THE INVESTIGATION .....	E-3
E1.4.1 <u>Study Location and Duration</u> .....	E-3
E1.4.2 <u>Collection Requirements and Available Environmental Data</u> .....	E-4
E1.5 COMPONENTS OF ECOLOGICAL RISK ASSESSMENT .....	E-5
<u>E2.0 SITE DESCRIPTION</u> .....	E-7
E2.1 PHYSICAL .....	E-7
E2.1.1 <u>Climate</u> .....	E-7
E2.1.2 <u>Physiography and Topography</u> .....	E-8
E2.1.3 <u>Geology and Soils</u> .....	E-8
E2.1.4 <u>Hydrology</u> .....	E-9
E2.2 BIOLOGICAL .....	E-11
E2.2.1 <u>Terrestrial Ecosystem</u> .....	E-11
E2.2.2 <u>Aquatic Ecosystem</u> .....	E-13
E2.2.3 <u>Important (Target) Species and Habitats</u> .....	E-13
E2.2.4 <u>Threatened and Endangered Species</u> .....	E-14
<u>E3.0 METHODOLOGIES FOR ECOLOGICAL EVALUATION</u> .....	E-15
E3.1 APPROACH FOR ECOLOGICAL RISK ASSESSMENT .....	E-15
E3.2 CONCEPTUAL MODEL .....	E-16
E3.2.1 <u>Components of Conceptual Model</u> .....	E-16
E3.2.2 <u>Release Mechanisms and Exposure Pathways</u> .....	E-16
E3.2.3 <u>Exposure to Contaminants Via the Food Web</u> .....	E-18
E3.3 DATA COLLECTION .....	E-21
E3.3.1 <u>Source of Ecosystem Data for Site</u> .....	E-21
E3.3.2 <u>Aquatic Toxicological Testing Procedures</u> .....	E-25
E3.3.3 <u>Tissue Sampling</u> .....	E-25
E3.3.4 <u>Uncertainty Analyses for Data Collections</u> .....	E-26
E3.4 DATA EVALUATIONS .....	E-27
E3.5 TOXICITY ASSESSMENT .....	E-27
E3.5.1 <u>Procedure for Screening Chemicals</u> .....	E-27
E3.5.2 <u>Potential Candidates as Contaminants of Concern</u> .....	E-30

# TABLE OF CONTENTS

(continued)

<u>Section</u>	<u>Page</u>
E3.5.3 <u>Selected Contaminants of Concern for OU1 EE</u> .....	E-32
E3.5.4 <u>Development of Toxicity Reference Values and Final Reference Values</u> .....	E-38
E3.5.5 <u>Uncertainty Analysis for Toxicity Assessment</u> .....	E-43
E3.6 EXPOSURE ASSESSMENT .....	E-43
E3.6.1 <u>Purpose of Exposure Assessment</u> .....	E-43
E3.6.2 <u>Uncertainty Analyses for Exposure Assessment</u> .....	E-45
E3.7 RISK CHARACTERIZATION .....	E-45
E3.7.1 <u>Use of Outputs from Toxicity and Exposure Assessment</u> .....	E-46
E3.7.2 <u>Quantification of Risks from Individual and Multiple Chemicals</u> ...	E-46
E3.7.3 <u>Uncertainty Analysis for Risk Characterization</u> .....	E-48
E3.8 METHODS FOR ECOLOGICAL COMPARISONS .....	E-49
E3.8.1 <u>Taxonomic Group and Trophic Level Comparisons</u> .....	E-49
E3.8.2 <u>Ecological Endpoints for Habitat Comparisons</u> .....	E-50
E4.0 <u>RESULTS</u> .....	E-51
E4.1 TOXICITY ASSESSMENT .....	E-51
E4.1.1 <u>Conceptual Model</u> .....	E-51
E4.1.2 <u>Identification of Contaminants of Concern</u> .....	E-51
E4.1.3 <u>Development of Reference Concentrations</u> .....	E-52
E4.2 EXPOSURE ASSESSMENT .....	E-52
E4.2.1 <u>Fate and Transport of Selected Metals at OU1</u> .....	E-53
E4.2.2 <u>Soils</u> .....	E-57
E4.2.3 <u>Surface Water</u> .....	E-58
E4.2.4 <u>Sediments</u> .....	E-61
E4.2.5 <u>Biota</u> .....	E-61
E4.3 ECOLOGICAL RISK CHARACTERIZATION .....	E-62
E4.3.1 <u>Risks from Individual COCs Chromium</u> .....	E-62
E4.3.2 <u>Combined Risks from Exposure Points</u> .....	E-67
E4.4 ECOLOGICAL COMPARISONS AT OU1 .....	E-69
E4.4.1 <u>Taxonomic Group Comparisons</u> .....	E-69
E4.4.2 <u>Food Web Comparisons</u> .....	E-70
E4.4.3 <u>Ecological Habitat Comparisons</u> .....	E-71

TABLE OF CONTENTS  
(continued)

<u>Section</u>	<u>Page</u>
E5.0 <u>SUMMARY</u> .....	E-74
E6.0 <u>CONCLUSIONS</u> .....	E-82
E7.0 <u>REFERENCES</u> .....	E-83

ATTACHMENT E.A POTENTIAL TOXICITY OF CONTAMINANTS OF CONCERN

ATTACHMENT E.B TISSUE ANALYSES

## LIST OF TABLES

### Table

E1.4-1	Sampling Design Matrix for OU1 Ecological Studies
E3.5-1	Chemicals Selected as Contaminants of Concern for the OU1 EE
E4.1-1	Toxicity Reference Values (TRVs) for OU1 Environmental Evaluation
E4.1-2	Final Reference Values (FRVs) for OU1 EE Contaminants of Concern: Surficial Soils
E4.1-3	Final Reference Values (FRVs) for Contaminants of Concern: Surface Water
E4.2-1	Species Used for Tissue Sampling
E4.2-2	Background Concentrations of OU1 Target Metals in Biological Tissues at RFP
E4.2-3	Vegetation Tissue Samples Exceeding Background at OU1 Sites
E4.2-4	Small Mammal Tissue Samples Exceeding Background at OU1 Sites
E4.2-5	Terrestrial Arthropod Tissue Samples Exceeding Background at OU1 Sites
E4.2-6	Fish Tissue Samples Exceeding Background at OU1 Sites
E4.3-1	Distribution of Contaminants of Concern Among Environmental Media at OU1
E4.3-2	Hazard Quotient Values for Chromium in OU1 IHSS Source Areas
E4.3-3	Hazard Quotient Values for Zinc in OU1 IHSS Source Areas
E4.3-4	Hazard Quotient Values for Lead in OU1 IHSS Source Areas
E4.4-1	Terrestrial and Aquatic Taxonomic Richness Percentage Comparison for OU1 EE
E4.4-2	Terrestrial and Aquatic Taxonomic Richness Statistical Comparison for OU1 EE
E4.4-3	Terrestrial and Aquatic Trophic Level Percentage Comparison for OU1 EE
E4.4-4	Terrestrial and Aquatic Trophic Level Statistical Comparison for OU1 EE
E4.4-5	Results for Plant and Animal Endpoints for OU1 EE

## LIST OF FIGURES

### Figure

- E1.4-1 General Location of Rocky Flats Plant
- E2.2-1 Operable Unit 1 Terrestrial Study Sites
- E2.2-2 Terrestrial Reference Sites
- E2.2-3 Aquatic Study Sites and Aquatic Reference Sites
- E3.1-1 Components of Ecological Risk Assessment at OU1
- E3.2-1 Conceptual Exposure Model for OU1 Ecological Risk Assessment
- E3.2-2 Generalized Food Web and Exposure Pathway of a Terrestrial Ecosystem with an Associated Aquatic System
- E3.2-3 Detailed Terrestrial Food Web, OU1
- E3.2-4 Detailed Aquatic Food Web, OU1
- E3.5-1 Screening Process to Identify Contaminants with Potential Ecological Impacts at OU1 within Rocky Flats Plant
- E3.5-2 Decision Process for Determination of Toxicity Reference Values (TRVs)
- E3.5-3 Decision Process for Determination of Final Reference Values (FRVs)
- E3.7-1 Relationship Between Environmental Risk Level and Hazard Quotient (HQ)
- E4.1-1 Screening Process for Soils to Identify Contaminants with Potential Ecological Impacts at OU1 within Rocky Flats Plant
- E4.1-2 Screening Process for Surface Waters to Identify Contaminants with Potential Ecological Impacts at OU1 within Rocky Flats Plant
- E4.1-3 Screening Process for Sediments to Identify Contaminants with Potential Ecological Impacts at OU1 within Rocky Flats Plant
- E4.2-1 Metals Exceeding Background in OU1 Surface Soils
- E4.2-2 Radionuclides Detected in OU1 Surface Soils
- E4.2-3 Summary of Contaminants of Concern in Surface Waters at OU1: Dissolved Metals and Radionuclides
- E4.2-4 Summary of Contaminants of Concern in Surface Water at OU1: Exceedence of Water Quality Standards
- E4.2-5 Results of Aquatic Toxicity Screening, Woman Creek-1991
- E4.2-6 Summary of Contaminants of Concern in Sediments at OU1: Metals and Radionuclides

## LIST OF ACRONYMS AND ABBREVIATIONS

ac	acre
ARARs	applicable or relevant and appropriate requirements
ATP	adenosine triphosphate
Be	beryllium
BRA	Baseline Risk Assessment
°C	degrees Celsius
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COC	contaminant of concern
DOE	U.S. Department of Energy
DQO	data quality objective
EC <sub>50</sub>	median effective concentration
EE	environmental evaluation
EPA	U.S. Environmental Protection Agency
°F	degrees Fahrenheit
FRV	final reference value
FSP	field sampling plan
ft	foot/feet
GI	gastrointestinal
ha	hectare
HI	Hazard Index
HQ	Hazard Quotient
IHSS	Individual Hazardous Substance Site
IRIS	Integrated Risk Information System
km	kilometer
LC <sub>50</sub>	median lethal concentration
LD <sub>50</sub>	median lethal dose
LOAEC	lowest observed adverse effects concentration
LOAEL	lowest observed adverse effects level
MATC	maximum allowable tissue concentration
µg/g	micrograms per gram
mg/kg	milligrams per kilogram
mi	mile
NOAEC	no observed adverse effects concentration
NOAEL	no observed adverse effects level
NPDES	National Pollutant Discharge Elimination System
OUI	Operable Unit No. 1
PHE	Public Health Evaluation
PPA	property protection area
RCRA	Resource Conservation and Recovery Act
RFI/RI	RCRA Facility Investigation/Remedial Investigation
RFP	Rocky Flats Plant

LIST OF ACRONYMS AND ABBREVIATIONS  
(continued)

SARA	Superfund Amendment and Reauthorization Act
SOP	Standard Operating Procedure
TRV	toxicity reference value

## GLOSSARY OF TERMS

absorption	the uptake of water or dissolved chemicals by a cell or an organism
acute	occurring over a short period of time; used to describe brief exposures
adhesion	steady or firm attachment
alluvium	clay, silt, sand, gravel, or similar detrital material deposited by running water
anthropogenic	of, relating to, or influenced by the impact of man on nature
arthropods	members of the phylum Arthropoda, which includes crustaceans, insects, and spiders
baseline	a measure of present conditions
benthic	organisms living on the bottom of water bodies
bioaccumulation	the increase of pollutant concentration passing into terrestrial species including intake from food and water
bioavailable	contaminants in a form readily incorporated into living tissue
bioconcentration	the increase of pollutant concentration from water when passing directly into aquatic species
biomagnification	the increase in concentration of pollutant in animal tissue in successive members of a food chain
caliche	a crust of calcium carbonate that forms on the stony soil of arid regions
carcinogen	a substance or agent capable of inducing cancer
carnivores	predatory organisms that eat animals
chronic	marked by long duration or frequent recurrence
coalescing	growing together



## GLOSSARY OF TERMS

(continued)

colluvium	a loose, heterogeneous deposit of soil and rock debris accumulated on a base of a slope and deposited by sheet wash and downslope creep
community	an assemblage of populations of different species within a specified location in space and time
consumer	an animal that feeds upon other organisms in a food chain
Cretaceous	a geologic time period that was the last of three periods in the Mesozoic era spanning the time 135 to 65 million years ago
cryptogam	a plant that reproduces by spores not flowers or seeds
demyelinate	to remove or destroy the outer sheath of nerve fibers
dermal	relating to skin and especially to the dermis layer
detritivore	an organism that feeds on dead organisms or partially decomposed organic matter
detritus	loose material (organic or inorganic in nature) that results directly from disintegration
eolian	pertaining to the wind; erosion and deposition accomplished by the wind
Median Effective Concentration (EC <sub>50</sub> )	The concentration of material estimated to be effective in producing some sublethal response in 50 percent of the test organisms.
ecological receptors	plant and animal species or groups of species exposed to contaminants
ecological risk assessment	the process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors

## GLOSSARY OF TERMS

(continued)

ecology	a branch of science concerned with the interrelationship of organisms and their environments
ecosystem	an ecological community, or group of communities, together with the physical environment, considered as a unit
ecotoxicology	the study of pollutants in ecosystems
endangered species	plant or animal species which are in danger of extinction
episodically	occurring, appearing, or changing at usually irregular intervals
fecundity	relative number of eggs, sperm, or young produced by an animal
food web	the totality of interacting food chains
Final Reference Value (FRV)	benchmark concentration of a chemical that is derived from TRV data and natural background levels (see Figure E3.5-3)
habitats	a place or type of environment where a plant or animal naturally or normally lives and grows
heavy metals	a metal, typically, of the transition group elements with a specific gravity of 5.0, or greater
hematocrit	volumes of corpuscles and fluids in blood
herbivore	any animal which relies chiefly or solely on vegetation for its food
hydric	of, or relating to an abundance of moisture
hydrostratigraphic unit	a body of rock having considerable lateral extent and composing "a geologic framework" for a reasonably distinct groundwater system
hypoplasia	a condition of arrested development in which an organ or part remains below the normal size or in an immature state

## GLOSSARY OF TERMS

(continued)

hypoxia	a deficiency of oxygen reaching the tissues of the body
ichthyofauna	the fish life of a particular region
imbibition	mechanical absorption of water by capillarity and other mechanisms, especially these processes in cellulose and other dead organic materials
interspecific variation	variation existing or arising between species
interstitial water	subsurface water in the voids of a rock layers
intraspecific variation	variation occurring within a species or involving members of one species
Median Lethal Dose (LD <sub>50</sub> )	the dose of material that is estimated to be lethal to 50 percent of the test organisms. Usually indicates the quantity of a material introduced directly into the body by injection or ingestion
Median Lethal Concentration (LC <sub>50</sub> )	the concentration of material in water to which test organisms are exposed that is estimated to be lethal to 50 percent of the test organisms. Usually expressed as a time-dependent value (e.g., 24-hour or 96-hour LC; the concentration estimated to be lethal to 50 percent of the test organisms after 24 or 96 hours of exposure)
Lowest Observed Adverse Effect Level	the lowest concentration in a medium (water, soil, sediment) which produces an observable adverse effect
Maximum Acceptable Toxicant (MATC)	the hypothetical toxic threshold concentration Concentration lying in a range bounded at the lower end by the highest tested concentration having no observed effect (NOEL) and at the higher end by the lowest tested concentration having a significant toxic effect (LOAEL) in a life cycle (full chronic) or partial life cycle (partial chronic) test.

## GLOSSARY OF TERMS

(continued)

mesic	of, or relating to a medium amount of moisture
mitosis	a process that takes place in the nucleus of a dividing cell, involves typically a series of steps consisting of prophase, metaphase, anaphase, and telophase, and results in the formation of two new nuclei each having the same number of chromosomes as the parent nucleus
mutagenic effect	a permanent alteration of the genetic material within living cells caused by a chemical or physical agent
nektonic	aquatic organism which actively swims (fish)
No Observable Adverse Effect Concentration (NOAEC)	the highest concentration in a medium which does not produce an observable adverse effect
omnivores	animals that eat both animal and vegetable materials
organic compounds	a compound containing carbon, especially as an essential component
periphyton	organisms that live attached to underwater surfaces
perturbation	to throw into confusion or disorder
phytoplankton	minute floating aquatic plants
phytotoxic	poisonous to plants
population	the total number of individuals of a species in a given area
primary producers	plants that can use sunlight as an energy source to produce carbohydrates
Quaternary	a geologic time period that was the second period of the Cenozoic era spanning the time 3 million years ago to the present

## GLOSSARY OF TERMS

(continued)

radionuclides	radioactive nuclides (a species of atom characterized by the number of neutrons and protons in its nucleus)
species richness	the total number of species in a given taxonomic category
standing crop	the total amount of plant tissue in a particular location at any given time
stressor	any physical, chemical, or biological entity that can induce an adverse response
subcrop	bedrock geological unit where surface is at the alluvium bedrock continuum
syncline	a trough of stratified rock of which the core contains the younger rocks; it is generally concave upward
target animals	those species most susceptible to contaminants, yet with enough biomass available to incorporate into a toxicological investigation
teratogenicity	tendency to cause developmental malformations and monstrosities
threatened species	a plant or animal species that is extremely rare, but not yet threatened with extinction
toxicity	a generic parameter designated to include concentrations of a material that have been observed to adversely effect a particular test organism.
transcuticular	transport across the cutical of plant tissue
trophic level	one of the hierarchical strata of a food web characterized by organisms which are the same number of steps removed from the primary producers
Toxicity Reference Value (TRV)	benchmark concentration of a chemical below that which sublethal toxic effects would be expected in most sensitive species (see Figure E3.5-2)

## GLOSSARY OF TERMS

(continued)

turbidity	the state, condition, or quality of opaqueness or reduced clarity of a fluid, due to the presence of suspended matter
vapor-phase (contaminants)	contaminants suspended in the air or in a gaseous state
xeric	of, or characterized by, or adapted to extremely dry habitat
zooplankton	minute floating animal life of a body of water

## Appendix E • Text

Environmental  
Evaluation

Phase III  
RFI/RI Report

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## EXECUTIVE SUMMARY

Under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and Resource Conservation and Recovery Act (RCRA) several operable units have been designated as areas at the Rocky Flats Plant (RFP) that require evaluation and remediation. Operable Unit No. 1 (OU1) is a designated area of the 881 Hillside for which a remedial investigation was required.

The purpose of the OU1 Remedial Investigation/Environmental Evaluation (RI/EE) is to answer two questions in sequence. First, are there contaminants of concern (COCs) capable of producing ecological risk within the OU1 area at RFP that are the result of plant releases? Second, are those COCs producing adverse ecological impacts? The EE assesses ecological risk for identified COCs that are above background levels and compares the structure and function of the ecosystem at the OU1 study area.

The principal objective of the EE at RFP is to collect data necessary to determine the nature, extent, distribution, and migration pathways of contaminants within the OU1 study area that have the potential to cause adverse ecological impacts.

The EE consists of an ecological risk assessment and ecological comparisons. A five-step process for the ecological risk assessment was used, while the ecological comparisons consisted of two components. The ecological risk assessment steps were data collected, data evaluation, toxicity assessment, exposure assessment, and risk characterization. The ecological evaluation consisted of species diversity and trophic level comparisons between the OU1 study area and Rock Creek watershed, the reference area.

The period of chemical and biological data collections at RFP for the OU1 EE was January 1991 through March 1992. A total of 139 biological tissue samples were collected to determine if COCs were bioavailable to the ecosystem. Ecological samples of plant and animal taxa, populations, and communities reported 399 taxa. There were 219 species of plants and 180 taxa



of animals within the OU1 study area. The most important factor affecting species diversity in communities at RFP is the amount of moisture available to support plant growth, the primary producers in the food web, and food for animals.

The toxicity assessment considers chemicals at OU1 that present a significant threat to ecological receptors and evaluates their potential toxicity. General toxicity information on each COC is used to develop toxicity and final reference values (TRVs and FRVs) for comparison with actual and estimated exposures at OU1. The first stage screening of COCs included heavy metals, cyanide, and radionuclides (because of their high profile at the site) that have been detected above background in soils, surface water, or sediments at OU1. Soils, surface water, and sediments are considered the main pathways for direct exposure of ecological receptors to contaminated media. The second stage screening process was based primarily on exceedance of RFP background concentrations, and secondarily on relative toxicity and bioavailability. The screening process identified chromium, lead, and zinc for soils; chromium, lead, and mercury for surface water; and no COCs were identified for sediments at OU1. Radionuclide levels in environmental media was of no concern to ecological receptors because of the very low concentrations.

The exposure assessment is conducted to estimate the magnitude of actual or predicted exposure concentrations and pathways by which ecological receptors are potentially exposed to contamination from the COCs occurring at Individual Hazardous Substance Sites (IHSSs) within the OU1 study area. Several very restricted areas have elevated concentrations of heavy metals in the soil. Most metals did not exceed background concentrations by more than twofold and probably do not represent contamination from releases at RFP. The background concentration of COCs at RFP was assumed to be below the toxicity threshold for metals of ecological receptors. Moreover, the exposure assessment failed to provide clear evidence of transport of contaminants away from the localized source areas at IHSSs. However, there was some evidence that sources in upgradient areas impacted water and sediment chemistry in areas adjacent to and

downgradient from OU1. The upgradient areas having the highest metal concentration were natural ground water seeps affected by the highly mineralized bedrock.

The risk characterization summarizes and combines outputs of the toxicity and exposure assessment sections to estimate the relative risk from exposure to COCs at OU1, both in quantitative expressions and qualitative statements. The risk characterization focuses on toxicological risks to ecological receptors from exposure to chromium, lead, zinc, and mercury, and the aggregate risk of simultaneous exposure to these COCs.

Chromium risks were assessed for soils and surface water. The Hazard Quotient (HQ) method indicated a moderate risk level associated with the highest chromium concentrations in IHSSs 119.1 and 119.2 for soils and low risk level for all other sites in the OU1 study area. Chromium was above background levels in surface water in only a single sample. No samples exceeded the (chronic) Colorado Surface Water Quality Standard for total chromium of 170 micrograms per liter ( $\mu\text{g/l}$ ). The restricted distribution of chromium at OU1 and lack of downgradient contamination indicate low risk to ecological receptors from exposure, and little or no risk of off-site transport.

Lead risks were assessed for soils and surface water. Lead was detected at concentrations above background in soils at four sites within and around IHSSs 119.1 and 119.2. Concentrations at three of the sites were approximately 10 percent above background. The fourth sample from IHSS 119.1 contained lead at 228 milligrams per kilogram ( $\text{mg/kg}$ ) (greater than fourfold the RFP background); a second sample from the same site contained only 78  $\text{mg/kg}$ . These data suggest that lead contamination in soils within IHSS 119.1 at OU1 is highly localized. The HQ method indicated low risk to ecological receptors from lead in soils at OU1 because of the restricted distribution. In surface water, lead was above background for Woman Creek upgradient and downgradient of the OU1 study area. The highest lead concentration in the Woman Creek drainage was 13.2  $\mu\text{g/l}$  from a tributary south of the plant site and upgradient of OU1 study area. The South Interceptor Ditch had the overall highest lead concentration, 38.4  $\mu\text{g/l}$  (dissolved);

however, lead levels of this magnitude were detected only once. The HQ value for the highest lead concentration in surface water from the single sample in the South Interceptor Ditch corresponds to high environmental risk. However, the overall risk to ecological receptors due to lead exposure in OU1 is considered low. This evaluation is based on the generally low level of lead in soils, surface water, and sediments, and the restricted distribution of areas with elevated concentrations. Lead concentration in biological tissue from OU1 indicate minimal uptake of lead.

Zinc risks were assessed for soils at OU1 where the distribution was similar to that of chromium. The average zinc concentration was near background with only three of 28 sample sites exceeding twofold the background concentration at RFP. The highest zinc concentrations were detected in IHSS 119.2 and around Building 881. None of the zinc HQ values for soil indicated risks above the low level. The overall HQ values for vegetation and soil invertebrates also are at the low risk level.

Mercury risks were assessed for surface water at OU1. Mercury concentrations did not exceed background in OU1 soils, surface water, or sediments in Woman Creek downgradient from OU1 IHSSs. Mercury concentration in surface water samples did exceed background in two sites on branches of Woman Creek that drain areas south of the RFP industrial area. However, these areas are outside potential impact from OU1 areas and the concentrations exceeded background by less than 30 percent. Mercury concentrations did exceed background in surface water samples at two stations in the South Interceptor Ditch; however, the source is likely upgradient of OU1 sources and the water from the South Interceptor Ditch, a ground water collection system, is not discharged into surface streams without treatment. The highest concentration of mercury in surface water from the single station in the South Interceptor Ditch was 1.0 µg/l; the HQ value indicates a moderate ecological risk. The risk from mercury in surface water at other stations is low. The acute and chronic Colorado Water Quality Standards for mercury are 2.4 and 0.1 µg/l, respectively. Therefore, the overall risk to ecological receptors from mercury exposure at OU1 is judged to be low. This assessment is due to the infrequent exceedances of background

levels in surface water and the lack of evidence for contamination in other media, including biological tissues collected from the site.

The common denominator in this report between the ecological risk characterization section and ecological comparisons section is the toxicity threshold for sensitive organisms (i.e., the concentration of chemicals that produce measurable toxic effects on ecological receptors). Since each species has a different sensitivity threshold, increasing the concentration of toxic chemicals selectively impacts intolerant species. This is reflected by reductions in species diversity (richness), as well as occasional increases in abundance of the remaining tolerant species because of reduced competition for food and habitat. Food webs respond in a similar manner by loss of species performing functions at impacted trophic levels. An ecosystem may become dysfunctional if chemical concentrations are such that species are eliminated.

Ecological comparisons at RFP were conducted at two organizational levels for the OU1 ecosystem—taxonomic structure and trophic function. As environmental stress from pollutants is gradually increased, the number of species decreases in response to surpassing toxic thresholds for individual species. Since each species occupies a niche (the ecological role or trophic function), then impacts are measurable also by trophic level comparisons. Together, the taxonomic and trophic level comparisons provide a yardstick to assess the health of an ecosystem.

The taxonomic group comparisons provided an estimate of the ecosystem's general health based on species diversity within the OU1 study area. The terrestrial ecosystem revealed no difference between the percentage of small mammalian species at OU1 and the Rock Creek reference area. The highest difference was only 3 percent, indicating similar species richness in the two areas. Small mammals are very sensitive indicators of stress caused by COCs entering the food pathway because they are primarily omnivores and herbivores and live in close contact with the soil, the major exposure point in the OU1 study area. The aquatic ecosystem showed slight differences in species richness for plankton and benthic macroinvertebrates. Rock Creek had 11 percent more plankton species and 9 percent less benthic macroinvertebrate taxa than the OU1 study area.

These differences were expected since the semiarid climate caused both streams to have recurring loss of habitat during intermittent flow.

The species used in the taxonomic level comparison were organized by trophic levels for a food web comparison. Terrestrial arthropods were compared separately since a less detailed identification endpoint was selected for study objectives. As in the taxonomic comparisons, trophic comparisons revealed much similarity between the OU1 and the Rock Creek ecosystems. The maximum percentage difference for any trophic level between these two areas was approximately 4 percent. Comparisons of terrestrial arthropods showed differences of only 6 percent or less between the two areas.

Trophic level comparisons for aquatic primary producers and omnivores showed differences up to 10 percent between OU1 streams and Rock Creek. The comparison of aquatic primary producers does not differ from the plankton comparison. Comparison of species richness for omnivores at the two areas revealed greater species richness at OU1, reflecting good ecosystem health in the OU1 study area.

Results for food web comparisons did not indicate a stressed ecosystem, nor did they reveal ecological problem areas. These results, along with those of tissue sample analysis, should be viewed as further weight-of-evidence indicating an ecologically healthy state at the OU1 study area.

## E1.0 INTRODUCTION

The Phase III Resource Conservation and Recovery Act (RCRA) Facility Investigation/ Comprehensive Environmental Response, Compensation, and Recovery Act (CERCLA) Remedial Investigation (RFI/RI) at Operable Unit No. 1 (OU1) 881 Hillside Area at the Rocky Flats Plant (RFP) includes a Baseline Risk Assessment (BRA). The BRA is comprised of the environmental evaluation (EE) and the Public Health Evaluation (PHE). Appendix E represents the EE portion of the BRA and addresses hazards to ecological receptors other than humans and domesticated animals (EPA 1989a).

### E1.1 PURPOSE

The purpose of the OU1 RI/EE is to answer two questions in sequence. First, are there contaminants of concern (COCs) capable of producing ecological risk within the OU1 area at RFP that are the result of plant releases? Second, are those COCs producing adverse ecological impacts? The EE provides an ecological risk assessment for identified COCs that are above background levels and compares the structure and function of the ecosystem at the OU1 study area. It is important to recognize that EEs are not research projects; they are not intended to prove cause and effect, nor are they designed to answer long-term research needs. Instead, an EE is an essential element in determining overall risk and protecting public health, welfare, and the environment. Regulatory agencies need to examine ecological effects and routes of exposure so that important impacts and transport pathways are not overlooked and reasonable estimates are made of health and environmental effects.

### E1.2 FOCUS OF OU1 ENVIRONMENTAL EVALUATIONS

#### E1.2.1 Guidelines Used for Structuring Environmental Evaluation

The framework used for this ecological risk assessment was taken from the following: Risk Assessment Guidance for Superfund Volume II Environmental Evaluation Manual (EPA 1989a); Supplemental Risk Assessment Guidance for the Superfund Program (EPA 1989d); Ecological Risk Assessment Methods: A Review and Evaluation of Past Practices in the Superfund and RCRA Programs (EPA 1989c); the draft version of Framework for Ecological Risk Assessment

(EPA 1992a), and Guidelines for Exposure Assessment (EPA 1992b). The primary purpose of using these guidelines was to produce a simple, flexible structure for conducting and evaluating potential ecological risks within OU1 at RFP.

#### E1.2.2 Definition and Concept of Ecological Risk Assessment

Ecological risk assessment is a procedure that estimates the possibility of adverse effects occurring in an ecosystem, or any part of an ecosystem, as a result of a perturbation. Regulatory mandates for ecological risk assessment are found in Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA [Superfund]), as amended by the Superfund Amendment and Reauthorization Act of 1986 (SARA) and other statutes designed to protect wildlife, fisheries, endangered and threatened species, and valued habitats.

The ecological risk assessment in this report was conducted using a three step process. The first step (screening) consisted of identifying potential COCs that are known to be present at OU1. The second step (characterizing potential impacts) consisted of reviewing research and regulatory findings to determine potential toxicity and behavior of COCs in the soils, surface waters, and sediments. The third step (actual ecological impacts) consisted of evaluating available ecological and chemical data from RFP to determine impacts resulting from the release of COCs at OU1.

#### **E1.3 OBJECTIVES**

The OU1 EE includes the three following general objectives:

- To determine background levels of potentially toxic substances
- To determine if potentially toxic substances are bioavailable in the OU1 study area
- To evaluate the risk of impact on ecological receptors from identified chemical conditions (ecological risk assessment)

The third objective, the ecological risk assessment, includes determining the extent or likelihood of measurable impacts and determining the potential of human activities, including remedial actions, to cause adverse ecological effects.

## E1.4 SCOPE OF THE INVESTIGATION

### E1.4.1 Study Location and Duration

The study site is located at RFP, a government-owned, contractor-operated nuclear facility located in northern Jefferson County, Colorado, approximately 25 kilometers (km) (16 miles [mi]) northwest of Denver (Figure E1.4-1). The period of data collection at RFP for the OU1 EE was January 1991 through March 1992.

The OU1 study area is located on the south side of the RFP security area in Woman Creek watershed. Woman Creek watershed drains approximately 1,144 hectares (ha) (2,826 acres [ac]) south of the industrial area and the east-west access road, and flows into Standley Lake, a domestic water supply reservoir.

The OU1 study area is south-facing, and slopes toward Woman Creek from the 881 Hillside area. Individual Hazardous Substance Sites (IHSSs) within the OU1 study area were designated as high priority because it is possible that COCs have been released at these sites based on historical accounts of use or accidental releases (Rockwell International 1987).

The following sites are designated as IHSSs at OU1:

- Oil Sludge Pit Site (IHSS 102)
- Chemical Burial Site (IHSS 103)
- Liquid Dumping Site (IHSS 104)
- Out-of-Service Fuel Oil Tank Sites (IHSS 105.1 and 105.2)
- Outfall Site (IHSS 106)
- Building 881 Hillside Oil Leak Site (IHSS 107)
- Multiple Solvent Spill Sites (IHSS 119.1 and 119.2)
- Radioactive Site – 800 Area Site No.1 (IHSS 130)
- Sanitary Waste Line Leak Site (IHSS 145)



A detailed description of the releases associated with each of the IHSSs in OU1 is contained in Section 1.2.2 of the RFI/RI report (Volumes I and II) and the Historical Release Report (DOE 1992b). Results of Phase III investigations to determine the extent of contamination are detailed in Section E4.0. Based on historical accounts and results of previous investigations, possible contaminants at OU1 include several organic compounds, radionuclides, and some heavy metals. The possible threat of these contaminants to ecological receptors was investigated during this EE. The first stage in the investigation was the identification of COCs, which is documented in the OU1 EE Field Sampling Plan (DOE 1991b). Based on criteria developed to select COCs, several heavy metals and a few radionuclides were identified for further investigation. The selection of COCs and criteria used to identify them are presented in Section E3.5.

The reference area for OU1 samples was located in the Rock Creek watershed. Samples from Rock Creek watershed were used to determine background values for COCs in soils, surface waters, sediments, biological tissue, and for comparison of ecological community endpoints. Rock Creek watershed drains approximately 629 ha (1,554 ac) in the northwestern portion of RFP and flows northeast. The channel length of Rock Creek on the RFP reservation is about 4 km (2.5 mi). This watershed is generally considered upwind of the plant industrial area, and is not hydrologically connected to it.

#### E1.4.2 Collection Requirements and Available Environmental Data

A matrix showing the types and number of samples collected for evaluating OU1 and the RFP sites, as adapted from the OU1 EE Work Plan, is shown in Table E1.4-1. These data were previously presented in the RFP Baseline Report (DOE 1992c); therefore, only summary information as required for the EE is presented in this report. Tissue sample data add new information for this evaluation. A total of 139 tissue samples were collected to provide data to evaluate OU1.

Data used in this report for EEs are located in the following reports:

- Phase II Geologic Characterization Data Acquisition Surface Geologic Mapping of the Rocky Flats Plant and Vicinity, Jefferson and Boulder Counties, Colorado (EG&G 1992)

- Baseline Biological Characterization of Terrestrial and Aquatic Habitats at the Rocky Flats Plant, Golden, Colorado (DOE 1992c)

## E1.5 COMPONENTS OF ECOLOGICAL RISK ASSESSMENT

There are five elements in the ecological risk assessment process:

- Data Collections
- Data Evaluation
- Toxicity Assessment
- Exposure Assessment
- Risk Characterization

Existing data were screened to determine if additional data collections were required. Data collection and evaluation involves gathering and analyzing the site data relevant to the EE and identifying the substances present at the site that are the focus of the risk assessment process.

Data collected specifically for the EE included the following:

- Background conditions for physical, chemical, and biological characteristics of the site
- Tissue samples of potentially contaminated species
- Species numbers, diversity, and richness

Field data were then evaluated to identify potential COCs. Data quality objectives (DQOs) were identified in accordance with U.S. Environmental Protection Agency (EPA) guidance (EPA 1987) and included identification and involvement of data users, development of a conceptual model, determination of data needs and uses, selection of analytical sampling options, and development of an overall sampling design. The toxicity assessment considers the type of adverse environmental effects associated with chemical exposures, the relationship between magnitude of exposure and adverse effects, and related uncertainties such as the weight of evidence of a particular chemical's toxicity to ecological receptors. The exposure assessment component of the EE identified exposure pathways and exposed populations. An exposure assessment links COCs to potential impacts on exposed populations. Concentrations of COCs found at OU1 were compared with standards for protecting the environment given by EPA documents and State of

Colorado codes (EPA 1992b; CCR 1989). Estimations of exposure concentrations and contaminant intake levels were made. The risk characterization summarizes and combines outputs of the toxicity assessment and exposure assessment information; it qualitatively describes and quantifies risks to exposed populations.

## E2.0 SITE DESCRIPTION

Administrative and production buildings at RFP are located within an approximately 156 ha (384 ac) industrial area protected by security fences. The remaining area, known as the property protection area (PPA), provides a buffer zone of approximately 2,496 ha (6,166 ac). The original purchase of land for RFP in 1951 included 1,020 ha (2,520 ac); in 1974 an additional 1,632 ha (4,030 ac) were acquired to expand the size of the PPA.

The original mission of RFP was the fabrication of nuclear weapon components from plutonium, uranium, and nonradioactive metals (principally beryllium and stainless steel). Parts made at RFP were shipped elsewhere for assembly. In addition, RFP reprocessed components removed from obsolete weapons to recover plutonium. This reprocessing generated radioactive, hazardous, and mixed (radioactive and hazardous) wastes. Current waste handling practices involve on-site and off-site recycling of hazardous materials, and on-site storage of the wastes.

Under CERCLA and RCRA, several operable units have been designated as areas at RFP that require evaluation and possible remediation. The Environmental Restoration Program Phase 1 Installation Assessment (DOE 1986) identified the 12 IHSSs within OU1 listed in Section E1.4.1. These areas are located on the hillside south and east of Building 881.

## E2.1 PHYSICAL

### E2.1.1 Climate

The area surrounding the RFP has a semiarid climate characteristic of much of the central Rocky Mountain region. Approximately 40 percent of the 38-centimeter (15-inch) annual precipitation falls during the spring season, much of it as snow. Thunderstorms (June to August) account for an additional 30 percent of the annual precipitation. Autumn and winter are drier seasons, accounting for 19 percent and 11 percent of the annual precipitation, respectively. Snowfall averages 216 centimeters (85 inches) per year, falling from October through May (USDA 1980). Temperatures are moderate; extremely hot and cold weather is usually of short duration. On average, daily summer temperatures range from 13 degrees Celsius (°C) to 29°C (55 degrees

Fahrenheit [°F] to 85°F), and winter temperatures range from -7°C to 7°C (20°F to 45°F). The low average relative humidity (46 percent) is due to the blocking effect of the Rocky Mountains.

The prevailing wind is from the northwest. The average wind speed in spring, the season of highest winds, is 16 kilometers per hour or 10 miles per hour. The diurnal drainage wind cycle of the Front Range area influences the local wind patterns at RFP. Specifically, Woman Creek receives and channels wind flow from the industrial area. Prevailing northwest winds can carry particulate contaminants downslope from OU1 and then eastward down the Woman Creek drainage.

#### E2.1.2 Physiography and Topography

The environment at RFP is influenced by the site's proximity to the Front Range of the Rocky Mountains, as well as its location on a broad, eastward sloping plain of coalescing alluvial fans. The elevation of RFP varies from 1,890 meters (m) (6,200 feet [ft]) at the western boundary to 1,722 m (5,650 ft) at the southeastern corner. The western terraces and the divides between the three creek drainages, including the ridge upon which the industrial area is located, are quite flat. Three intermittent streams flow eastward across the site. The creek drainages vary from moderate slopes in lower Woman Creek and Walnut Creek to quite steep in upper Rock Creek. Rock Creek's steeper ravines have a southwest to northeast orientation while the other two creeks have wider valleys that trend west to east.

#### E2.1.3 Geology and Soils

The bedrock geology, surficial geology, and soils of OU1 combine to influence both the hydrology and the vegetation, and thus the ecology of the OU1 area. A detailed description of the geology of the OU1 area is found in the OU1 Phase III RFI/RI Work Plan (DOE 1991a) and Section 3.6 of the Phase III RFI/RI report.

### Bedrock Geology

The bedrock at RFP is made up of Cretaceous sedimentary rocks of the Arapahoe Formation, Laramie Formation, and Fox Hills Sandstone. The bedrock of the OU1 area is part of the Laramie Formation which is comprised of an upper claystone member and a lower sandstone member. The upper claystone member is 90- to 150-m (300- to 500-ft) thick and the lower sandstone member is 90-m (300-ft) thick. The lower Laramie Formation and the Fox Hills Sandstone subcrop in the clay and gravel pits west of the industrial area.

### Surficial Geology

Surficial materials from which the soils of OU1 were formed consist of Quaternary and Recent valley fill alluvium, alluvial fan deposits of the Rocky Flats Alluvium, and colluvium (EG&G 1992).

### Soils

The soils at RFP are moderately deep, well-drained clay, cobbly clay, and sandy loams with moderate to low permeability. Soils of the terraces and the upper hillsides, where gravel and cobbles are common, are represented by combinations of Denver and Kutch series. These mesic Torriertic Argiustolls are sandy loams formed from Rocky Flats Alluvium. Lower hillsides and areas toward the eastern boundary of RFP have soils from the Standley, Nunn, and Valmont series which are largely mesic Ardic Argiustolls. Bottomland soils are largely stratified loamy alluvium, made up of mesic Ustic Torrifluvents from the Haverson series. Runoff from these soils is generally rapid, and erosion can be severe on the steep hillsides (USDA 1980).

## E2.1.4 Hydrology

### Ground Water

As defined in the Final Ground Water Assessment Plan for Rocky Flats (DOE 1992a), Rocky Flats Alluvium, valley fill alluvium, colluvium, bedrock sandstones, and weathered claystones of the Arapahoe and Laramie Formations comprise the uppermost hydrostratigraphic unit at RFP. In general, ground water of OU1 is confined to bedrock depressions during low water conditions,

with isolated pockets of water located near seeps. Ground water levels at RFP rise annually in response to spring recharge and decline the remainder of the year.

### Surface Water

Three headwater streams of the South Platte River drainage basin flow generally from west to east across RFP. These watersheds, shown in Figure E2.2-2, are Woman Creek, Walnut Creek and Rock Creek (in order of occurrence from south to north). Woman Creek and Walnut Creek are tributaries of Big Dry Creek. Rock Creek flows into the drainage system composed of Coal Creek and then St. Vrain Creek. Portions of the Woman Creek and Rock Creek watersheds were surveyed for the OU1 EE.

Woman Creek – The Woman Creek watershed includes the area south of the industrial area and the east-west access road, and flows into Standley Lake, a water supply reservoir that discharges into Big Dry Creek. The channel length of Woman Creek on the RFP reservation is about 5 kilometers (3.1 miles). There are several impoundments in the Woman Creek watershed. One stormwater detention pond, Pond C-1, is located in the Woman Creek channel; one catchment and detention pond, Pond C-2, is located at the end of the South Interceptor Ditch. The South Interceptor Ditch and Pond C-2 system collect and store surface water runoff and some ground water from the industrial area. While this catchment system is within the Woman Creek drainage area, it was designed as a closed system with no surface discharge into Woman Creek. Water collected in Pond C-2 is pumped via an above ground pipeline to either Pond A-4 or B-5 in Walnut Creek. An emergency option exists for pumping water from Pond C-2 to the Broomfield Diversion Ditch.

Rock Creek – Rock Creek watershed drains the northwestern portion of RFP. The length of the channel that crosses RFP is about 4 km (2.5 mi). There are several small ponds in the Rock Creek watershed. Lindsay Pond is a stock pond constructed in the channel of Rock Creek on the former Lindsay Ranch property. One small stock pond near the north boundary is filled seasonally by runoff. One flooded clay and gravel pit located on the terraces west of the

industrial area also holds ponded water. This deep pit, which is not expected to have surface discharges, may function to collect some surface flow from the surrounding areas and ground-water discharges.

## E2.2 BIOLOGICAL

There were 219 species of plants and 180 animal taxa identified within the OU1 study area. Of the animal taxa, 26 percent are arthropods, 14 percent are birds, 4 percent are mammals, and 2 percent are herptiles.

### E2.2.1 Terrestrial Ecosystem

The majority of the plant species at OU1 contributing to the terrestrial communities belong to two groups—vascular cryptogams (2 species) and vascular plants (217 species). A complete list of all plant species documented at RFP is supplied in Appendix B of the Baseline Biological Characterization of Terrestrial and Aquatic Habitats at the Rocky Flats Plant (DOE 1992c). Terrestrial sample sites for OU1 and the reference area are depicted in Figures E2.2-1 and E2.2-2. Among the dominant vascular plants, various growth forms are represented. Trees and shrubs constituted 6 percent of the total number of species, forbs (broad-leaf herbs) 66 percent, graminoids (grasses and grass-like plants) 25 percent, and cactus the remaining 2 percent.

The flora of the entire RFP site are widely diverse due to varied geography, but reclamation activities (re-seeding) in the OU1 study area have limited the vegetation diversity of OU1. The OU1 study area comprises 4 percent of the total area of RFP. Although thirteen vegetative habitats are represented in OU1, two grassland habitats (mesic mixed grassland and reclaimed) are dominant, representing about 82 percent of the total area. Another 9 percent of the area is either developed or disturbed. Marsh habitats (tall marsh, short marsh, and open water) occupy about 4 percent, woodland habitat (primarily riparian) constitutes another 4 percent, and shrub habitats (short and bottomland shrub) account for the remaining 1 percent of the OU1 study area.



The taxa of animals identified at OU1 and at RFP as a whole, respectively, is 103 versus 124 families of arthropods, 1 versus 4 species of amphibians, 5 versus 8 species of reptiles, 56 versus 142 species of birds and 15 versus 32 species of mammals.

The arthropods, particularly the insect herbivores, are the most abundant terrestrial animal group. Arthropods serve as links in the food chain and make the energy produced by plants available to the secondary consumers. Passerine birds (songbirds) and small mammals are also abundant animal groups. Most animals in these two groups are herbivores or insectivores with a small number of omnivores. They control the number of insects, spread plant seeds, and serve as prey base for top carnivores such as raptors and coyotes.

Wildlife species at RFP are typical of those in similar habitats throughout the foothills because of the absence of barriers between the western plains and the surrounding foothill terrain. Wildlife habitat at RFP is characterized according to plant communities upon which wildlife depend for food and shelter, as outlined in the Baseline Report (DOE 1992c).

Various animals occupy OU1 habitats. Bull snakes, rattlesnakes and occasionally racers occur in many habitats, and western plains garter snakes were found in moist areas. Common birds include western meadowlarks, horned larks, morning doves, vesper sparrows, house finches, marsh hawks, red-tailed hawks, ferruginous hawks, rough-legged hawks, and great horned owls. Mallards and Canada geese breed on some of the small ponds. Medium-sized mammals are represented primarily by desert cottontails and muskrats, with a few black-tailed jackrabbits, white-tailed jackrabbits, and porcupines. Carnivores are primarily coyote, striped skunk, and raccoons. The most common large mammal is the mule deer. The Baseline Biological Characterization of Terrestrial and Aquatic Habitats at the Rocky Flats Plant (DOE 1992c) contains complete listings of all species by habitat type.

### E2.2.2 Aquatic Ecosystem

The aquatic ecosystem at OU1 includes two major habitat types: streams and ponds. Neither is well developed due to the semiarid climate and seasonal distribution of rainfall that occurs along the Front Range of the Rocky Mountains. The Woman Creek channel west of Pond C-1 and east of Pond C-2 is essentially in native condition. The ponds represent significant alteration of the natural drainage. As a result of limited and inconsistent surface water supplies, aquatic species with short life-cycles and smaller habitat requirements, such as benthic macroinvertebrates, have developed more diverse communities than fish. Fish are limited by intermittent streamflow, water temperature fluctuations, food, and habitat. During the annual low rainfall periods, habitat availability in the intermittent reaches of the Woman Creek watershed within the OU1 study area limits the number of life forms in the aquatic ecosystem. Aquatic samples were collected from OU1 and reference area sites depicted in Figure E2.2-3.

Benthic sampling in OU1 yielded 131 taxa of benthic macroinvertebrates, as compared with 155 taxa found sitewide. The highest densities of individuals were for the orders Oligochaeta (aquatic worms), Diptera (aquatic flies and midges) and Ephemeroptera (mayflies). All of these benthic organisms constitute food for fish and waterfowl.

The limited number of fish species (7) found in OU1 include stoneroller, fathead minnow, golden shiner, creek chub, white sucker, green sunfish and largemouth bass. Creek chub and fathead minnow are the most widely distributed. Carp and goldfish, which occur elsewhere in RFP, are not present in OU1.

### E2.2.3 Important (Target) Species and Habitats

At the initiation of study for OU1, a list of target species was prepared, consisting of 1 invertebrate, 1 fish, 5 upland game birds, 44 waterfowl, 27 raptors, 11 mid-sized mammals, 12 carnivores, and 4 large mammals. These were considered to be of importance because of their recognized status (state or federal), or for social or economic reasons. The list was reduced to include only arthropods, aquatic species, and small mammals because their home ranges were

small enough that they would be likely to spend their entire life cycle within the OU1 area. Also, these organisms were determined to be at high risk because of intimate contact with potentially contaminated soils or surface water. These target species are also important members of the food web and serve as possible vectors for biomagnification.

#### E2.2.4 Threatened and Endangered Species

Species of federally or state listed threatened, endangered or candidate animals and plants potentially present at OU1 were identified. The only one actually found within the OU1 study area, at site MR02A, was Preble's meadow jumping mouse (*Zapus hudsonius preblei*), occupying riparian habitat adjacent to Woman Creek. This species is a candidate for listing on the endangered species list, thereby indicating that the U.S. Fish and Wildlife Service considers it to be vulnerable.

### E3.0 METHODOLOGIES FOR ECOLOGICAL EVALUATION

There are two components for the EE at OU1. First, an ecological risk assessment procedure has been developed from EPA guidance documents (EPA 1989a,d). The ecological risk assessment evaluates expected impacts to ecological receptors from known concentrations of COCs at OU1. Second, ecological endpoint comparisons quantify differences between terrestrial and aquatic populations, communities, and habitats for OU1 and the reference area in Rock Creek watershed. The ecological endpoints were assessed using data collected for the RFP Baseline Report (DOE 1992c) and OU1 Field Sampling Plan (DOE 1991b). Data evaluations from these sampling programs previously were incorporated in the Baseline Report. Therefore, this EE presents only data that are necessary for ecological endpoint comparisons and food web analyses at OU1.

#### E3.1 APPROACH FOR ECOLOGICAL RISK ASSESSMENT

In accordance with the Framework for Ecological Risk Assessment (EPA 1992a) assessment of ecological risks on the OU1 site involved the following:

- Description of risks in terms of assessment endpoint
- Discussion of ecological significance of effects
- Summarization of overall confidence in the assessment
- Discussion of results with the risk manager

There are five steps for this ecological risk assessment. They are data collections, data evaluations, toxicity assessment, exposure assessment, and risk characterization (Figure E3.1-1). The steps are used to assess potential ecological risk from identified contaminants, if any, for three media—soil, surface water, and sediments. The risks are focused on potential adverse affects from exposure pathways at OU1. A conceptual model has been developed to identify and prioritize by significance the exposure pathways.

## E3.2 CONCEPTUAL MODEL

### E3.2.1 Components of Conceptual Model

Vegetation, wildlife, and aquatic organisms (plants and animals) can be exposed to contaminants directly through contact with contaminated media (soil, sediment, and water). Animals can also be indirectly exposed through consumption of contaminated forage or prey (Figure E3.2-1). The conceptual model was developed to identify exposure pathways and exposure points. Each exposure pathway consists of four elements: (1) source of contaminant, (2) mechanism of retention or transport medium, (3) an exposure route (e.g., ingestion), and (4) a receptor (EPA 1989a). These components can be further defined as involving primary or secondary sources and release mechanisms. A contaminant that has been released to the environment can be a contaminant source for other media. For example, soil contaminated by a previous spill could be a contaminant source for ground water or surface water.

### E3.2.2 Release Mechanisms and Exposure Pathways

A general conceptual model for OU1 was described in the Phase III RFI/RI Work Plan (DOE 1991a). The primary contaminant sources included in this model were the contaminated soils in the OU1 IHSSs and buried waste. The primary release mechanisms important to the EE are surface runoff, fugitive dust, and biotic uptake. Infiltration and percolation of contaminants into deep ground water may also have occurred. However, exposure due to ground-water contamination is not considered in this EE because deep ground-water movement at the site is highly limited, therefore, contaminants are not likely to reach surface water sources. Data on water chemistry from Woman Creek support this assumption. Furthermore, an interim remedial action directed at intercepting and treating ground water has been implemented since this study began. Air dispersal is not a major exposure pathway to wildlife because inhalation of contaminated particulates occurs episodically, and most species are too short-lived for this low level of exposure to have a significant effect on individuals. Species that are larger and longer-lived (e.g., coyotes and mule deer) are more mobile and thus spend a smaller proportion of their lives within the area of contamination.

Primary and secondary transport can result in a greatly expanded area and potential for exposure of biotic receptors (Figure E3.2-1). The most important abiotic media—soil, surface water, and sediments—act both as sources of direct exposure to a variety of plant and animal groups and as entry points for contaminant movement into the food web (Figure E3.2-3). Soil contaminants sufficiently mobile to be taken up through the foliage or roots can be distributed throughout the plant or differentially concentrated in certain tissues. The latter type of bioaccumulation is a key element in trophic exposure pathways. Some plant species are known to concentrate certain metals at levels toxic to wildlife. Contaminants that are phytotoxic can cause community effects by reducing or eliminating one or more plant species. In most cases, the concentrations in soil required to impact vegetation would not be expected outside the OU1 source areas.

The most important release mechanism for dispersal of contaminants at OU1 is expected to be surface water runoff. Sheet runoff and rill erosion can cause transport of contaminants to soils outside the IHSS. Surface runoff is also important because Woman Creek is generally a low-energy stream, and contaminants attached to soil or sediment particles may accumulate in pools and detention ponds. During periods of high flow, dissolved or sediment-borne contaminants can be transported significant distances downstream. The small areal extent and closed nature of aquatic ecosystems result in continuous exposure of resident organisms; many terrestrial species rely heavily on surface water or aquatic prey. Direct exposure to contaminated surface water is a potential exposure pathway for both terrestrial and aquatic species. Terrestrial vertebrates may imbibe substantial quantities of water. Incidental ingestion of water while feeding on aquatic prey is potentially important for species such as raccoons, great blue herons, and ducks.

Direct exposure to contaminated soil is also a main pathway of concern for wildlife. Many principal prey species spend much of their lives in intimate contact with surficial soils. This can include larvae or adults of many invertebrates, as well as small vertebrates such as rodents, which may ingest substantial quantities of dirt while burrowing, grooming, or feeding. Ingestion is unrelated to mobility of the contaminant. Dermal absorption is not an important route of direct

exposure for OU1 COCs. Ingestion of contaminated soil is of less concern for deer and birds, primarily because they are more wide-ranging and spend less time in contact with the soil.

Bioaccumulation of a chemical is another pathway that can result in toxic exposure, even when the ambient concentrations are relatively nontoxic. In general, bioaccumulation is limited relative to persistent organic pollutants such as chlorinated organic pesticides. Unless organo-metallic forms are present, the potential for bioaccumulation of most metals is limited to bioconcentration directly from environmental media or to organisms in lower trophic levels. However, many metals can bioaccumulate in aquatic systems because the main mechanism of exposure is through absorption directly from water across gills and other external surfaces.

Biomagnification, the successive accumulation of a pollutant with increasing trophic level, is an important mechanism of bioaccumulation in terrestrial systems since most species obtain the majority of contaminant loads from food sources. However, this process is not as prevalent for metals as it is for pesticides due generally to decreasing bioavailability (Martin and Coughtrey 1982; Moriarty 1983).

### E3.2.3 Exposure to Contaminants Via the Food Web

Contaminants physically or chemically taken up by biota may be further distributed via the food web. Members of all trophic (feeding) levels may also come in direct contact with contaminated media; most of the feeding relationships ultimately lead to predatory vertebrates; and terrestrial and aquatic components are interconnected.

#### Terrestrial Food Web

The generalized food web reflects that plants take nutrients, moisture, and (potentially) contaminants from the soil and incorporate them into their tissue (Figure E3.2-2). Plant tissue is eaten by herbivores (both vertebrate and invertebrate), which in turn are consumed by predators (loosely referred to as carnivores). Omnivores consume both plant and animal tissue. The food

web eventually leads to large (top) predators such as coyotes and raptors. At each step in the terrestrial food web, an organism ingests any contaminants that its forage or prey has incorporated into its tissue, as well as residual soil on its surface. This could include dust on leaves and dirt on the pelage of burrowing rodents. The total amount of exposure increases from one trophic level to the next. However, where contaminant sources are patchy, as at RFP, the larger home range of higher-level predators has the effect of reducing their overall exposure. Only for COCs that bioaccumulate within plant or animal tissue, is biomagnification between trophic levels apt to be significant.

The three most important terrestrial trophic pathways for potential exposure to contaminants in the Woman Creek ecosystem are the vascular plant, arthropod, and herbivorous mammal pathways (Figure E3.2-3). These pathways are most important at OU1 because of interactions previously described in Section E3.2.2.

#### Aquatic Food Web

The aquatic food web in the OU1 area includes organisms associated with Woman Creek and Ponds C-1 and C-2. The base of the food web is comprised of sediment, organic detritus, and water (Figure E3.2-4). The sediment and detritus provide a substrate and source of nutrients for rooted aquatic plants such as cattails and coyote willows. Phytoplankton absorb nutrients and contaminants directly from the water column. Macroinvertebrate fauna, ranging from small chironomid or caddisfly larvae to crayfish, feed on detritus, plankton, and periphyton. These organisms are important prey for fish.

Aquatic food webs differ from terrestrial food webs with regard to contaminant exposure in that dissolved contaminants may be absorbed directly from the water. A combination of bioconcentration from water, ingestion of contaminated prey, and the generally restricted ranges of aquatic organisms can lead to significant bioaccumulation. Even compounds that readily adsorb onto sediment can impact aquatic biota during periods of turbidity (e.g., spring runoff).



They can also be introduced into the food web by detritivorous invertebrates or bottom-feeding fishes.

As in terrestrial ecosystems, higher-level predators tend to be the receptors of greatest concern because of trophic biomagnification and greater longevity. This is exacerbated where aquatic and terrestrial food webs interconnect, because the aquatic predators may in turn be prey for terrestrial predators. The following animals are potentially important links between aquatic and terrestrial ecosystems in the OU1 study area (Figure E3.2-4):

- Wading birds or waterbirds consume fish; herons are of greatest concern because of their longevity and fidelity to specific feeding areas
- Ducks that consume aquatic plants or macroinvertebrates are in turn consumed by coyotes, foxes or owls
- Raccoons consume a variety of aquatic vertebrates and macroinvertebrates (especially crayfish)
- Muskrats consume aquatic as well as nearby terrestrial vegetation and may be preyed upon by terrestrial carnivores such as coyotes or large raptors

Flow of energy and contaminants from terrestrial to aquatic ecosystems can also occur when insects or other invertebrates land on the water or are washed in following a rain. These terrestrial organisms may be fed upon by fish, adult amphibians, or detritivores. However, the amount of biomass and contaminants potentially exchanged by this route is very small compared to the flow from aquatic to terrestrial organisms.

#### Synopsis of Exposure Pathways

The potentially most significant exposure pathways of OU1 COCs to biota may be summarized as follows:

- Direct exposure of receptors to soil contaminants within OU1 IHSSs, and outside the IHSS areas
- Direct exposure of aquatic organisms to contaminants transported into surface water by wind, runoff, or shallow ground water

- Imbibition of contaminated surface water (including seeps and springs) by terrestrial vertebrates
- Consumption of contaminated plant material by herbivores
- Consumption of contaminated animal tissue by predators

Data collected during the Phase III RFI/RI and ongoing RFP monitoring programs was used to evaluate exposure to contaminants in abiotic media. Evaluation of contaminant uptake by plants and animals was carried out by comparison of tissue samples from OU1 with samples from areas upgradient of OU1 and from reference areas.

### E3.3 DATA COLLECTION

Initial field efforts in the OU1 EE study included mapping vegetation communities and making a general survey of terrestrial and aquatic habitats available at RFP to aid in development of a sampling strategy. Based on the initial surveys, the extent of the OU1 study area was established, and the location and extent of the reference area was selected. The OU1 EE was conducted in accordance with the Draft Final Phase III RFI/RI Environmental Evaluation Work Plan: Field Sampling Plan, 881 Hillside, Operable Unit No. 1 (DOE 1991b) and EMAD Standard Operating Procedures (SOP) Manual (DOE 1991c).

#### E3.3.1 Source of Ecosystem Data for Site

Survey sites in both OU1 and a reference (control) area were used to determine whether contamination resulting from activities in OU1 have, or could in the future adversely affect ecological health. The reference area was used to provide specimens unlikely to be contaminated for comparison with OU1 specimens.

The physical area of OU1 was expanded to include downwind and downdrainage areas. This expanded area, designated as the OU1 study area, allowed for examination of the continuum of potential contamination levels. The design allowed sampling of a variety of habitats in a potentially affected zone downdrainage and downwind from 881 buildings. The study area

included OU1, the 881 Hillside, and areas outside the industrial area boundary fence extending west to the southeast PPA access road, to the southern levee of Woman Creek, and east to include Pond C-2. Woman Creek formerly received surface water runoff from the industrial area, but construction of the South Interceptor Ditch between the industrial area and Woman Creek has diverted surface water flow to Pond C-2. Woman Creek may potentially be affected by groundwater seepage, windblown materials, and overflow from the South Interceptor Ditch.

Construction of the french drain between the 881 Hillside and the South Interceptor Ditch has changed the character of a portion of the study area, and has also altered the water flow patterns of the area.

Criteria for selection of the reference area included location upwind and updrainage from 881 Hillside area activities and away from all other known RFP activities with potential to produce contamination, habitats as close to natural conditions as possible, and an area unimpacted by other local industrial activities. The northwest portion of the PPA, the Rock Creek watershed, met these criteria.

After study and reference areas were delineated, the terrestrial habitats, as identified in the SOPs, present within these areas were identified. Specific sample sites for terrestrial animal species were established within these habitats. Because of their concurrency, the OU1 EE was designed to take advantage of the database formed during the baseline biological characterization of terrestrial and aquatic habitats (DOE 1992c). The locations of terrestrial sample sites in the study area and reference area are shown in Figures E2.2-1 and E2.2-2, respectively.

Study sites for the aquatic ecosystem were selected from stream and pond habitats in the Rock Creek and Woman Creek watersheds. Locations upstream from the study area on Woman Creek and locations on Rock Creek were used as reference sites. Study area sites were selected along Woman Creek downstream of OU1 and along the South Interceptor Ditch, including Pond C-2 (Figure E2.2-3).

### Terrestrial Sampling Procedures

Biotic diversity and community composition reflect the health of an ecosystem. Species present in either terrestrial or aquatic ecosystems can indicate the degree of stress on a community due to perturbations, as pollution intolerant species are under-represented in a stressed environment. The sampling program was designed to detect environmental stress through comparisons between study and reference areas.

Sampling for each ecological component was conducted in accordance with the ecology SOPs. The sampling matrix for all taxa sampled is presented in Table E1.4-1. Detailed site layout descriptions can be found in the Field Sampling Plan (DOE 1991b) and the appropriate SOP (DOE 1991c). The primary objective was to collect data for comparison between reference and study area sites that would reveal any adverse impacts in the study area.

### Plants

Vegetation was mapped in accordance with SOP EE.11 (DOE 1991c), which defined the habitats present at RFP. Crown cover of herbaceous plants, determined by the point-intercept method (SOP EE.10), was recorded by species. These data were used to calculate the percentage of plant cover, bare ground, rock, and litter. Belt transects were used to record species presence at each vegetation sampling site. Species density was determined for trees, cacti, yucca, and shrubs. Species richness was determined by the number of species occurring within each of the belt transects. Production (standing crop) sampling included all herbaceous species (grasses, forbs, and subshrubs) encountered within the sample quadrat (a half-meter square frame).

### Animals

Relative abundance surveys designed to document the relative numbers of all observed species in each sample area were used for amphibians, reptiles, medium-sized mammals, carnivores, and large mammals. Arthropods were collected by sweepnetting, in accordance with SOP EE.9 (DOE 1991c). Amphibians and reptiles were sampled in accordance with SOP EE.8 (DOE 1991c). Small mammal sampling was collected in accordance with SOP EE.6 (DOE 1991c). No specific

sampling programs were designed for carnivores; however, these species were recorded during relative abundance surveys. Sampling of large mammals was in accordance with SOP EE.5 (DOE 1991c). Relative abundance surveys were the primary data collection tool for large mammals. Casual observations and winter season counts were also added to data collected (see DOE 1992c for further information).

#### Data Analyses

Terrestrial plant sampling data analysis included calculation of areal extent of plant communities (mapped and quantified), and evaluation of species richness for use in the OUI trophic level evaluation. Background ecological evaluations on species, populations, and communities is found in the Baseline Report (DOE 1992c).

#### Aquatic Sampling Procedures

Phytoplankton samples were collected during the late summer from study and reference area impoundments (ponds) in accordance with SOP EE.3 (DOE 1991c). Periphyton were collected during late summer. Artificial substrates (tiles and diatomers) were used as required in accordance with SOP EE.1 (DOE 1991c). Benthic macroinvertebrates were collected from streams and impoundments in accordance with SOP EE.2 (DOE 1991c). These organisms were collected during May-June and August-September. Study and reference area aquatic sites were evaluated for the likelihood that fish species were present. Fish were sampled in May-June and in August-September according to the most appropriate method as outlined in SOP EE.4 (DOE 1991c).

#### Data Analyses

In addition to preparation of a taxonomic list, species richness, relative abundance, relative density, and biomass (expressed as ash-free dry weight and biovolume for phytoplankton only) were recorded. Species density was recorded as a relative measure because of the magnitude of ordinary variance among samples as a result of substrate dissimilarities between samples at the

same station. Relative numbers provided a mechanism for comparison between study and reference sites to determine community health.

#### E3.3.2 Aquatic Toxicological Testing Procedures

Acute aquatic toxicity screens were conducted on samples collected from Woman Creek in order to ascertain gross toxicity of surface water and determine whether any toxicity detected could be a result of contaminants originating from the OU1 area. To do this, samples were collected from sites on Woman Creek upgradient of OU1, and downgradient of OU1. Samples were also collected from Pond C-2, which is located on the historic Woman Creek channel but currently is isolated from Woman Creek surface flows by a diversion. Flow from the South Interceptor Ditch is the principal source of surface water flow into Pond C-2 at present.

Samples were collected during low flow in August 1991 in accordance with SOP 4.2, Surface Water Sampling (DOE 1991c) and the instructions and protocols from the toxicity testing laboratory. Samples were immediately placed in a cooler with blue ice and transported to the laboratory within 6 hours of collection. Toxicity tests commenced within 24 hours of collection and were conducted following the techniques described in Peltier and Weber (1985) using fathead minnows and water-fleas as test organisms. These procedures are consistent with the Colorado Department of Health/Water Quality Control Commission and EPA Region III guidelines for biomonitoring. Hardness, alkalinity, conductivity, ammonia, pH, and dissolved oxygen were measured in samples prior to the toxicity tests. Other water chemistry data were obtained from results of RFP monthly surface water sampling activities.

#### E3.3.3 Tissue Sampling

Tissue samples were composed of plant and animal groups considered to be vulnerable components of the ecosystem (i.e., animals with small home ranges with intimate contact with the soil, plants, and aquatic organisms). Samples were taken from all sites where possible. Groups collected for tissue analysis of the terrestrial system included vascular plants, grasshoppers, small mammals, and reptiles. Specimens were collected from crayfish,

salamanders, and fish for analysis of potential risk to the aquatic system. Procedures for collection and preparation followed the Field Sampling Plan (DOE 1991b) and the appropriate SOP (DOE 1991c).

#### E3.3.4 Uncertainty Analyses for Data Collections

All data collection methods, whether ecological or toxicological, have limitations because ecosystems are dynamic. In ecological sampling, qualitative methods and timing may have excluded or under-represented certain groups. Relative abundance surveys such as those conducted for this project do not produce a quantitative measure of species present. Instead, they were conducted to assess habitat use for the more mobile organisms. Budget and time limitations precluded the inclusion of quantitative surveys. Also, the lack of nocturnal surveys may have missed or at least under-estimated abundance of nocturnal species, such as owls and jackrabbits.

Gross estimates of biological uptake of contaminants was deemed adequate for this first phase of analysis, so the gastrointestinal (GI) tract and fur (mammals) of the specimens were not separated. Inclusion of the GI tract introduces variability in the results due to potential ingestion of contaminated soil that is not bioavailable as well as differences in the length of time since last feeding. Animals which had just eaten could have higher contaminant levels than those which had an empty GI tract, and contaminants present in animals with a full GI tract might be passed through the animal's system without being assimilated.

Other limitations that may have affected the toxicity data were a low number of replicates and analysis of COCs. For example, laboratory analysis of some elements or compounds were reported as total levels, while only a specific form of the compound is toxic to the biota. Thus, high levels in the abiotic media may or may not signify a problem for the biota. Low number of replicates resulted from the need to composite many individuals from a site to make weight requirements requested by the laboratory. (Field weight of 50 grams minimum of tissue was required for analysis, but small species do not weigh this much.) This problem was compounded

in the aquatic sampling by limited fish populations due to the intermittent nature of the streams. A low number of replicates makes estimation of true mean difficult.

#### E3.4 DATA EVALUATIONS

The process of identifying COCs that would be analyzed in the risk characterization (Section E3.7) was an ever narrowing process. Contaminants were eliminated based on magnitude above background levels, actual toxicity (low, moderate, or high), and level of bioavailability. An example of the screening process is presented in Figure E3.5-1.

#### E3.5 TOXICITY ASSESSMENT

This section summarizes potential toxicity of COCs for OU1 and the potential pathways by which ecological receptors may be exposed to COCs. General toxicological information on each COC was used to develop toxicological reference values for comparison with actual and estimated exposures at OU1. A conceptual model for exposure of ecological receptors to COCs in the OU1 study area is presented in Section E3.2. Concentrations of COCs were measured or estimated for each exposure point identified in the conceptual model. Relative hazard of these exposures was assessed by comparison with existing regulatory standards, toxicity information on COCs, and geochemical background concentrations at RFP. The procedure for developing reference values is presented in Section E3.5.4. Information upon which the reference values were based is in the Toxicity Summary section of Appendix E.B.

##### E3.5.1 Procedure for Screening Chemicals

Contaminants of concern (COCs) are chemicals that result from activities at a hazardous waste site, are suspected to occur in environmental media as a result of activities at the site, and have the potential to damage natural populations or ecosystems. Identification of chemicals as COCs provides a focus for further investigation of the threat to ecological receptors at the site. The COCs are evaluated for their potential toxicity at the concentrations found in environmental media at the site, their potential for transport away from the site, and the potential for bioaccumulation.



As noted in Section E1.0, the focus of the OU1 EE was to determine if chemicals were released as a result of plant activities in OU1, and if these chemicals have adversely impacted the environment. Identification of COCs for the OU1 EE was based on three basic criteria: (1) documentation of occurrence of the chemical in environmental media, (2) ecotoxicity of the chemical, and (3) the extent of contamination at RFP. These criteria are discussed in more detail below.

### Occurrence

The known or suspected occurrence of a chemical in environmental media was gleaned from the following sources:

- Existing data from abiotic media (soil, water, sediment)
- Waste stream identification and disposal practices
- Process analyses to identify potentially hazardous substances used in large quantities
- Historical accounts of use or accidental releases

The resulting list of chemicals was then evaluated for ecotoxicity and extent of contamination at the site.

### Ecotoxicity

For purposes of evaluating potential COCs, ecotoxicity of a chemical was determined from its documented adverse effects on biota and its synergistic effects with other chemicals. A chemical was considered for inclusion in the list of COCs if, at levels detected within the operable unit, it exhibits any of the following characteristics:

- Acute and chronic toxicity, including mortality and teratogenicity
- Sublethal toxicity, including carcinogenicity, reduced growth rates, reduced fecundity, and behavioral effects
- Toxicity resulting from bioaccumulation due to absorption of the chemical directly from environmental media or ingestion of contaminated food items

The above information was extracted from federal or state regulatory guidelines, chemical information databases, or the open scientific literature. The resulting list of chemicals was then evaluated for extent of contamination at RFP.

#### Extent of Contamination

To support identification of a chemical as a COC, the extent of its contamination must be such that it results in significant exposure to ecological receptors. A chemical was retained in the list of COCs if it was present above natural background concentrations and exhibited one or both of the following characteristics:

- Present above regulatory standards, or above applicable or relevant and appropriate requirements (ARARs)
- Present above risk-based acceptable levels

The chemical was finally identified as a COC if it also was reported in greater than 5 percent of the soil, surface water, or sediment samples analyzed from OU1 and exhibited at least one of the following characteristics:

- Widely distributed
- Occurred in ecologically sensitive areas such as marshes or seeps that might serve as a drinking water source for wildlife
- Occurred in localized areas of high concentration ("hot spots")

Widely distributed is defined as occurrence of a chemical that is not restricted to one sample site. For OU1, a chemical was categorized as widely distributed if the number of borings in which it was found (hits) comprised at least 20 percent of the total borings analyzed for the chemical.

#### Additional Factors

Depending on physical and chemical properties, contaminants may become unequally distributed among environmental media or among components within a medium. The result may be

differential bioavailability or contaminant exposure of species or populations. Factors affecting distribution in environmental media include, but are not limited to, the following:

- Persistence – resistance to degradation by abiotic or biotic processes
- Volatility – tendency to vaporize, thus reducing soil or water concentration
- Mobility – the degree to which a chemical tends to migrate within or between environmental media, putting further resources at risk
- Solubility – tendency to dissolve in aqueous media (which may affect mobility in surface water and ground water) and to segregate into soil or sediment
- Differential accumulation – tendency to segregate into different environmental media or components of a single medium (e.g., adsorption affinity)

#### E3.5.2 Potential Candidates as Contaminants of Concern

COCs for the OU1 EE were identified in two stages. An initial list of COCs was developed from data collected during the Phase I and Phase II RFI/RIs and presented in the Phase III RFI/RI Work Plan (DOE 1990a). Using the criteria described above, 20 chemicals were originally identified as COCs (Table E3.5-1). Identification was based primarily on concentrations of chemicals in soils within OU1 IHSSs. Most of the suspected releases occurred at least several years before this investigation, and the primary sources of contaminants have long since been removed. Therefore, residual contamination of soils in IHSSs represent a secondary source for direct exposure to soils within the IHSS, or for secondary releases if contaminants are transported away from the IHSSs. Identification of COCs was also based on concentrations of chemicals in surface water and sediments of the South Interceptor Ditch and Woman Creek. Due to erosion, chemicals in soils within a drainage basin tend to accumulate in sediments of waterways draining the basin, potentially magnifying the concentrations to which aquatic organisms are exposed.

Fourteen metals were identified as COCs because concentrations in OU1 soils and/or surface water exceeded RFP background concentrations as presented in the Background Geochemical Characterization Report (DOE 1990b). Several radionuclides were also identified because concentrations exceeded RFP backgrounds. Known concentrations of radionuclides were well

below those known to cause acute or effects in ecological receptors. However, the radionuclides were included as COCs because of their importance in the PHE. Contamination of OU1 ground water with organic solvents was also suspected. However, no organic chemicals were included in the COCs because organic contaminants were restricted to deep (greater than 15 feet), relatively immobile ground water and therefore are relatively unavailable for exposure to ecological receptors. Preliminary evaluation of the data from Phase III investigations was focused on the concentrations of these chemicals in soils and surface water. This initial list also provided a basis for selection of target analytes for analysis of biological samples collected during Phase III field operations.

Results of the Phase III investigations were then used to further focus the investigation on those COCs that present significant environmental risk at OU1. This subsequent evaluation was made on the basis of (1) Phase III data on surficial and deep soils; (2) redefined RFP background concentrations; (3) the relative toxicity of the chemical at concentrations detected during Phase III operations; and (4) assessment of the probability that the presence of a chemical resulted from activities at OU1. This secondary evaluation of COCs, summarized in Figure E3.5-1, also represents the initial stages of risk characterization.

A major consideration in the assessment of the COCs was the high natural background concentrations of metals and radionuclides associated with RFP soils. Use of this background information is important because geologic materials underlying the RFP area are naturally high in many of the metals named on the COC list for OU1. Under these conditions, evolutionary or physiological processes may have allowed local flora and fauna to become adapted or acclimatized to high ambient metal concentrations. Alternatively, the ambient conditions may naturally limit the structure of the ecological community at RFP.

For the purposes of risk assessment, it was assumed that species native to RFP exist at ambient metal concentrations well within their range of tolerance. If the chemical concentration did not exceed twofold the RFP background for a given environmental medium, the risk due to site

activity was considered low, and the chemical was excluded from the COCs for that medium. If the concentration exceeded the final reference value (FRV), the method of assessing the risk due to exposure is described in Section E3.7.

#### E3.5.3 Selected Contaminants of Concern for OU1 EE

The rationale for selecting the COCs carried forward into the risk analysis is provided below, along with information regarding their toxicity and behavior in environmental media. The radionuclides are all members of the actinide group; therefore, information on those five appears together in the radionuclide section following the metals. The 20 selected COCs are listed in Table E3.5-1.

Aluminum – Aluminum was present at concentrations above background levels established for RFP in both sediment and soil and met all the criteria for extent of contamination. This metal has been shown to have toxic effects on soil communities and on higher plants (Bartlett and Riego 1972; Runge 1984; Horst 1985) where effects are manifested as growth inhibition (Taylor et al. 1991). Aluminum is mobilized in terrestrial and aquatic ecosystems by increasing the acidity of soil and water. Studies in humans have shown that aluminum compounds can affect absorption of other elements in the GI tract and altered intestinal function (Goyer 1986). Aluminum is acutely toxic to fish in high concentrations, and sublethal effects in laboratory-tested fish include behavioral modifications, reduced growth rate, and developmental impairment (EPA 1988). Both toxic and sublethal effects have been demonstrated in benthic macroinvertebrates as well.

Arsenic – Arsenic was found at concentrations exceeding background. This metal exhibits two inorganic forms with differing toxic properties. Arsenic III binds to sulfhydryl groups on proteins, disrupting their function. Arsenic V is thought to selectively uncouple oxidative phosphorylation, poisoning aerobic ATP generation (i.e., energy production) (Fowler et al. 1977; Schiller et al. 1977). In plants, arsenites (arsenic III) are more toxic than arsenates (arsenic V). The former cause wilting, the latter cause chlorosis, and both are toxic if absorbed in sufficient

quantity. The distribution of arsenic through the food chain is greatly limited by its phytotoxic effects (i.e., plant injury would generally occur before concentrations toxic to wildlife could be reached). Arsenic is potentially toxic to all terrestrial invertebrates. Little information is available on the effects, toxicity, and potential for accumulation of arsenic in terrestrial vertebrates. However, lower forms of aquatic life (e.g., aquatic macroinvertebrates) may exhibit acute arsenic toxicity (EPA 1985a). Furthermore, laboratory studies showed acute toxicity of arsenic III to common freshwater fish and sublethal effects of reduced growth rate, behavioral alterations, decreased enzyme activity, alterations in blood chemistry, and decreased hematocrit (EPA 1985a). Arsenic may also inhibit DNA repair systems; at lower concentrations, it may have a mutagenic effect (Jernelov et al. 1978). The wide spectrum toxicity of arsenic stems from its ability to block the citric acid cycle, a basic metabolic pathway of all higher organisms. Arsenic is not usually bioconcentrated (EPA 1985a). Unlike many other heavy metals, the toxicity of arsenic III in aquatic animals appears to be independent of water hardness.

Beryllium – Beryllium was found at the OU1 site in concentrations exceeding background levels in surface water and sediment and is therefore available to aquatic receptors and terrestrial animals through ingestion. Beryllium met all of the criteria on extent of contamination except it is not above a pertinent biota ARAR. This metal is classified as hazardous by EPA. The toxicity of beryllium to aquatic biota is dependent on pH, alkalinity, and hardness of the water. It can be acutely poisonous to fish at high concentrations; but because of its low solubility in water, it is usually unavailable to most aquatic organisms. Bioaccumulation is therefore not important in determining its aquatic fate (Wilbur 1980). The primary vector for beryllium bioaccumulation in animals is inhalation rather than ingestion.

Cadmium – Cadmium was found at the OU1 site in concentrations exceeding background only in soil and is therefore absent in ecologically sensitive aquatic areas at concentrations of concern. Cadmium is thought to be one of the most toxic elements for plants causing reductions in growth rate (Taylor et al. 1991). It also adversely affects soil microflora, subsequently affecting higher vegetation. Cadmium has been shown to have a higher potential for concentration than any other

metal in most terrestrial invertebrates. Wildlife are exposed to cadmium primarily via ingestion of contaminated food and drinking water and in some situations, from aerial deposition (Beyer et al. 1985). Birds and mammals appear to be less sensitive to cadmium than are aquatic organisms. Cadmium accumulates in the liver and kidneys of vertebrates (Anderson and Van Hook 1973; Johnson et al. 1978). Cadmium is seen at higher concentrations in insectivores such as shrews than in herbivores such as field voles (Roberts and Johnson 1978; Scanlon 1979). Sublethal effects of cadmium in birds and mammals include reduced growth rate, anemia, hypoplasia in bone marrow and gonads, enlarged heart and behavioral changes.

Chromium – Chromium was found at the OU1 site in concentrations exceeding background in soil and sediment. Chromium was not above the biota standard based on EPA action criteria, but met all other criteria for extent of contamination. Chromium is naturally abundant and occurs in many oxidation states, but only trivalent and hexavalent forms are biologically toxic. No biomagnification of chromium has been observed in food chains; concentrations are usually highest at the lowest trophic levels (Eisler 1986). However, chromium does bioaccumulate and exhibits acute or chronic and sublethal effects on biota. Exposure to high concentrations in plants results in weight decrease. The response to chromium in terrestrial invertebrates is based on few data, but chromium appears not to concentrate. In aquatic fauna, hexavalent chromium adversely affects invertebrates by reducing survival and fecundity. Sublethal effects have also been observed in freshwater fish.

Copper – Copper was found at OU1 sites in concentrations exceeding background only in sediments. Copper meets the three ecotoxicity criteria, but its concentrations did not exceed the biota standard based on EPA action criteria and it did not occur in areas of locally high concentration. The bioavailability of copper depends on pH, redox potential, sediment type, water hardness and organic content. Copper is toxic to aquatic life with toxicity decreasing as alkalinity increases. Free cupric ions are more toxic than most organic and inorganic complexes. Fish and invertebrate species seem to be about equally sensitive to the chronic toxicity of copper (EPA 1985d).

Iron – Iron was found at OU1 sites in concentrations exceeding background in soil and sediment. Iron met all the criteria for extent of contamination, except it does not occur in areas of locally high concentration. Iron does not have a biota standard based on EPA action criteria. Iron exhibits acute or chronic and sublethal effects on biota. In terrestrial ecosystems, iron is typically bound to soil particles. Increased soil acidity may release potentially toxic amounts for uptake (Jernelov et al. 1978; Donahue et al. 1983) although soils are moderately alkaline at RFP. No literature was found on iron toxicity in terrestrial vertebrates, but based on studies with humans it is expected that ingestion of excessive amounts of iron would result in acute toxicity (Goyer 1986).

Lead – Lead was found at OU1 sites in concentrations exceeding background in soil only. Lead interferes with the activity of ATP-ase, an important metabolic enzyme, and is therefore potentially toxic to all organisms (Jernelov et al. 1978). Plants absorb lead from the soil via roots and from airborne dust on leaf surfaces. Lead inhibits plant growth, reduces photosynthesis, and reduces mitosis and water absorption (Demayo et al. 1982). Uptake of lead by plants is limited by low bioavailability of lead in soils (Eisler 1988). In studies of terrestrial invertebrates near mine sites, a range of invertebrate species bore very high body burdens of lead without showing any signs of toxicity. The toxicity of lead to mammals, however, is widely recognized. Mammalian toxicity stems from the tendency of lead to demyelinate nerve axons. High doses of lead induce abortion and increase skeletal malformations.

Manganese – Manganese was found in levels exceeding background in soil and sediment. It met all the criteria for extent of contamination, except it is not widely distributed. Manganese does not have a biota standard based on EPA action criteria. Manganese is relatively insoluble in basic soils but may be soluble in strongly acid soils, resulting in toxicity (NAS 1973). It is not clear whether manganese is bioaccumulated by plants and transported through the food chain at a toxic level. Manganese does not appear to be biomagnified between trophic levels. Toxic effects from high body burdens of manganese have not been observed in terrestrial invertebrates. There is little information on the toxicity and effects of manganese in terrestrial vertebrates,



although in humans large doses of manganese salts cause gastrointestinal irritation. In the aquatic environment, high concentrations of manganese have been reported to be acutely toxic to eels and rainbow trout in laboratory tests. The acute toxicity of manganese decreases with increased water hardness.

Mercury – Mercury concentrations were detected above background in surface water, soil, and sediment. Mercury meets all of three criteria for ecotoxicity. It met four of the five criteria for extent of contamination. Mercury does not have a biota standard based on EPA action criteria. Both terrestrial and aquatic plants accumulate traces of mercury, the amount depending on the species, location and chemical form of mercury available. The toxicity of mercury in animals is related to effects on membrane function, and toxicity is therefore widespread through all phyla. In addition, mercury is readily concentrated up the food chain. Because of the ability of various biota to methylate elemental mercury into more toxic and bioavailable forms, the presence of mercury has serious implications for ecosystems.

Silver – Silver occurred in concentrations exceeding background in sediments only, making it available to the aquatic ecosystem. It met all the criteria for extent of contamination except that it is below its biota standard based on EPA action criteria. Water hardness and chloride ion concentration are the two factors involved with acute silver toxicity in aquatic organisms; silver is more toxic in soft water. Silver bioaccumulates to some degree in aquatic food chains with bioconcentration factors ranging from less than 1 for bluegills to 240 for some benthic macroinvertebrates (EPA 1980).

Zinc – Zinc was found in concentrations exceeding background in all three abiotic media. It exceeded the biota standard based on EPA action criteria, but was not widely distributed. Zinc exhibits acute or chronic effects, sublethal effects, and bioaccumulation in biota. Zinc enters the food chain through aerial deposition on foliage or through uptake by plant roots. Although zinc is extremely soluble, uptake by roots is limited. Levels of zinc in excess of the micronutrient requirements of plants may result in reduction in growth rates. In aquatic systems, acute toxicity

to fish includes gill destruction and hypoxia. Exposure of fish to sublethal concentrations causes edema and necrosis of liver tissue (Rand and Petrocelli 1985). Toxicity decreases with increasing water hardness. Bioconcentration and transfer of zinc through both terrestrial and aquatic food chains has been documented.

Radionuclides – Actinide elements include number 89 (actinium) and higher numbered elements. At OU1 radionuclides of this series includes the following COCs: Americium-241, plutonium-239, radium-226, strontium-90, and total uranium. All of these were below the standards set for protection of human health and ecological receptors. At the extremely low levels that these radionuclides occur in the OU1 study area, no impacts are expected. In general, actinide nuclides form comparatively insoluble compounds in the environment and therefore are not considered biologically mobile. Transport in ecosystems is largely the result of erosion and leaching. Ecological receptors are affected primarily through inhalation and ingestion.

Terrestrial plant uptake from soil is generally considered to be low, especially for plutonium. Plant to soil concentration ratios for true uptake runs about  $10^{-4}$  or less for the oxide and hydroxide forms which usually occur in the environment. There is some evidence that somewhat higher plant to soil concentration ratios exist for uranium and americium. The actinides on vegetation frequently attach to the surfaces to greater extent than biological incorporation. Important to food pathways is that assimilation of all actinide elements from the gastrointestinal tract is assumed to be less than 0.01 percent (ICRP 1960).

Inorganics – Cyanide was found at OU1 sites in concentrations exceeding background in soils only. It is also absent in areas of locally high concentration, but met all other criteria on extent of contamination. Cyanide exhibits acute or chronic and sublethal effects on biota. While many chemical forms of cyanide exist in the environment, free cyanide is the primary toxic agent. No reports were found of cyanide biomagnification or cycling in living organisms, probably owing to its rapid detoxication. Cyanide seldom persists in soil owing to complexation, microbial metabolism, and loss from volatilization (Eisler 1991). In higher plants, elevated cyanide

concentrations inhibit respiration and ATP production, eventually leading to death (Towill et al. 1978). At lower concentrations, cyanide inhibits germination and growth. Single large exposures of cyanide are extremely lethal to terrestrial vertebrates, but sublethal doses, especially in diets, can be tolerated by many species for extended periods of time and perhaps indefinitely.

#### E3.5.4 Development of Toxicity Reference Values and Final Reference Values

The evaluation of ecological risks associated with contamination at OU1 was, in part, carried out using the HQ method (EPA 1989e). This method uses the ratio of the actual or estimated exposure concentrations to toxicologically based benchmark or reference values. In human health risk assessments the reference values, called reference doses, are based on toxicity to a single species, humans, and are readily available from EPA databases (i.e., Integrated Risk Information System [IRIS]) and medical literature.

The HQ method or modified versions of it have also been applied in ecological risk assessments (EPA 1989a; CDH 1990; EPA 1992a, 1992c). However, formal reference values are not readily available for most animal and plant species and must be derived from various sources. This section describes the process by which reference values were derived for use in this EE.

Two types of reference values were developed. Toxicity reference values (TRVs) (CDH 1990) were developed for exposure of major taxonomic groups to each chemical on the initial list of COCs. Data for TRV development were derived from regulatory standards and guidance and scientific literature on environmental toxicology. The TRV was chosen to represent the "no-observed adverse effects concentration" (NOAEC) for exposure of sensitive species to a given toxin. A final reference value (FRV) was developed using the TRV and RFP background concentrations. The FRV was then used to calculate the HQ and to characterize ecological risks. The following sources of information were used to develop reference values:

- Colorado State Water Quality Standards
- EPA Ambient Water Quality Criteria

- EPA or EPA-sponsored on-line databases such as IRIS and AQUIRE
- Open scientific literature concerning toxicity and bioaccumulation of the chemicals in question

### Toxicity Reference Values

The first step in selection of TRVs was to gather information on the toxicity of each of the COCs to six major taxonomic groups: vegetation, terrestrial invertebrates, small mammals, birds, aquatic invertebrates, and fish. This toxicity information is summarized in Appendix E.B.

The data used to develop TRVs were prioritized as outlined below:

- Regulatory standard or Ambient Water Quality Criteria (aquatic taxa only)
- Formally derived data relating to concentrations causing important sublethal effects such as the LOAEC, NOAEC, MATC, and EC<sub>50</sub>
- When formal data as above were not available, less well defined values for concentrations causing sublethal effects were used
- Formally derived median lethal exposures such as the median lethal dose (LD<sub>50</sub>), median lethal concentration (LC<sub>50</sub>), etc.
- Less well defined concentrations causing mortality

The procedure employed to select TRVs included steps intended to account for the possible uncertainty introduced by use of different types and sources of data. Safety factors were applied to avoid possible underestimation of toxicity. The procedure is inherently conservative in that sublethal effects were used when available, data were used for the most sensitive species noted in the literature, and safety factors were applied to the values to account for the sources of variation noted above. The method follows rationale presented by Lewis et al. (1990) and Fordham and Reagan (1991). Each source of uncertainty and the procedure for including estimates in the development of the TRV are summarized below. The overall process for identifying TRVs is depicted in Figure E3.5-2.

Uncertainty results when extrapolating toxicity information from a specific study to general applicability. Several sources of uncertainty and various means of accounting for uncertainty in setting regulatory standards or estimating hazards have been suggested (Dourson and Stara 1983; EPA 1985a, 1986, 1989a, 1989b; Lewis et al. 1990). Major sources of uncertainty include intraspecific variation, interspecific variation, extrapolation from laboratory results to field data, and differences among field sites. In addition, the applicability of data extracted from the literature depends upon the type of result presented and the methods used to arrive at the results. The type of result reported may be a formally defined toxicological endpoint such as a  $LD_{50}$  or LOAEC, or a less stringently defined measure of mortality or sublethal effect. Also considered is the probability that an effect was actually caused by the agent in question, or can be ascribed to other causes (Lewis et al. 1990).

The toxicity of many chemicals is known to depend on the conditions of exposure. For example, the toxicity of many metals to aquatic organisms is highly dependent upon pH, water hardness, and total organic carbon content of water. Conditions under which the studies reviewed were conducted were highly variable as were the toxic concentrations reported. Consequently, the application of results from a particular study to another site introduces some uncertainty into results and conclusions. To counter some of this uncertainty, the lowest toxic value encountered for the taxon was used.

Safety factors were applied to toxicity information derived from the literature to account for intraspecific variation in sensitivity to toxins. The safety factors described are based on empirical observations from many studies in which the actual relationships among statistically derived toxicity parameters were evaluated (Lewis et al. 1990). This approach was used to estimate the "no-observed adverse effects level" (NOAEL) when this parameter was not available for exposure of a given species to a given chemical. Available LOAECs were reduced by a factor of 3.5, which was the average LOAEC to NOAEC ratio for 27 terrestrial species (Weil and McCollister 1963). When concentrations causing an effect were defined as an  $EC_{50}$  or similar value, or when effective concentrations were not formally defined, the lowest concentration having an effect was

divided by 5, as application of this factor approximated the NOAEC in 96 percent of cases studied for laboratory mammals (Weil and McCollister 1963). When median lethal exposures such as an LD<sub>50</sub> or LC<sub>50</sub> were used, the concentration was reduced by a factor of 6 (Weil 1972; Lewis et al. 1990). When lethal exposures were presented, but no formal toxicological endpoint was derived, the lowest concentration showing lethality was also reduced by a factor of 6. This procedure provides protection to the most sensitive organisms in the environment; therefore, impacts to the ecosystem, communities, and populations are unlikely at this reduced concentration.

Interspecific variation in sensitivity represents the most important source of error in environmental risk assessment but may also be the most difficult to determine. For example, for a group of 12 fish species, the MATC for cadmium exposure in ambient water differed by a factor of 6 between the most sensitive and most resistant species (Rand and Petrocelli 1985). Similar values are available for other metals listed as COCs for OU1.

Uncertainty due to interspecific variation was countered in two ways. For each taxon, the toxicity values for the most sensitive species encountered were used as the base value. For most taxonomic groups this selection overestimated the sensitivity of the most resistant species by at least a factor of 5, and usually more than a factor of 10. Where possible, the toxicity values were chosen for species within the same genus or family as species found at RFP. In most cases, however, the literature was sparse and examples could be found only within the class or order of taxa occurring at RFP. When comparable toxicity values were available for fewer than five families, the toxicity value was reduced by a factor of 2 based on the assumption that the lowest toxic values found represent the sensitive end of the toxicity spectrum for a given taxon. If values were available for five or more families, the lowest value was used. Information on toxicity of COCs to aquatic invertebrates and fish was treated as recommended by EPA (EPA 1985a) and applied in the Ambient Water Quality Criteria (EPA 1980; 1985b,c,d,e,f; 1986; 1987a; 1988).

Colorado Water Quality Standards and EPA Ambient Water Quality Criteria were used without modification in development of surface water TRVs (CCR 1989). These standards are generally derived to protect the most sensitive species tested. Colorado regulatory standards have been promulgated for each of the heavy metals listed as COCs and for the radionuclides cesium-134, plutonium-238,-239,-240, radium-226,-228, strontium-90, and tritium. Values reported are for Class 1 warm water streams. Woman Creek and its tributaries at RFP may be classified as Class 2 stream segments because of low and intermittent flows. Water quality standards for Class 2 streams are set on a case-by-case basis by the Colorado Department of Health Water Quality Control Commission, which has not classified Woman Creek or its tributaries as Class 2 stream segments, nor have site-specific water quality standards been established. The values listed were taken from Table III of Colorado Water Quality Standards 3.1.0 (5 CCR 1002-8) as amended September 30, 1989 (CCR 1989). Values for cadmium, chromium III, copper, lead, silver, and zinc are dependent on hardness (calcium carbonate content) of the water body in question. Surface water TRVs were calculated using a hardness of 80, as this value is representative the lower hardness values encountered at RFP (and metals are less toxic in harder water).

#### Final Reference Values

The process for selecting the FRVs from TRVs and RFP background information is depicted in Figure E3.5-3. Briefly, the TRV or RFP background, whichever was greater, was used as the FRV. As noted previously, the geologic materials underlying the RFP area are naturally high in many of the metals named on the COC list for OU1 and background concentrations for several metals exceed the TRV. In such cases, the TRV probably overestimates the toxicity of the COC to RFP species, so the background concentration was used as an approximation of the NOAEL. In some cases, no reliable toxicological information was available for a chemical in a given medium. In such cases, exceedance of RFP background was noted, but no FRV was established.

In some cases the FRV for a chemical is equal to the RFP background. This may seem to conflict with the twofold background criterion for screening COCs. However, it should be stressed that the FRV is not an action level, merely a benchmark concentration for evaluating the

potential hazard to ecological receptors at RFP. Use of the FRV and background concentrations in risk characterization is discussed in Section E3.7. FRVs are presented in Section E4.1.1.

#### E3.5.5 Uncertainty Analysis for Toxicity Assessment

The uncertainty associated with identification of COCs include reliability of data on chemical concentration in environmental media, and the reliability of historical information on contaminant sources and location of contaminated areas. The uncertainty associated with these sources is not quantifiable. However, Phase III investigations were used to better define distribution of potential contaminants and to more accurately define the COCs that present an ecological threat at OU1. The sources of uncertainty in applying toxicity information and mechanisms used to counter the uncertainty are discussed above.

### **E3.6 EXPOSURE ASSESSMENT**

#### E3.6.1 Purpose of Exposure Assessment

The purpose of the exposure assessment is to evaluate actual or predicted exposure of identified ecological receptors to contamination resulting from IHSSs within OU1. Potential exposures were estimated based on concentrations of contaminants measured in soils, surface water, sediments, and biota collected from the study area and background areas.

The data used in the exposure assessment were drawn from the following sources:

- Operable Unit 1 Phase III RFI/RI Workplan (DOE 1991a)
- Phase III Site Characterization (Section 4.0 of the main RFI/RI report)
- Phase III Ecological Characterization (Section E2.0)
- Results of analysis of biological tissues collected during the Phase III investigation

Data on exposure of receptors to potential contamination in soils, surface water, and sediments are summarized in Section E4.2. Data presented in Section 4.0 of the RFI/RI report are not



represented here. Rather, data are summarized for those sites with samples exceeding the RFP background, regulatory standards, and/or reference values.

#### Methodology for Soils

COC concentration in soils collected during Phase III operations and RFP background were measured as total COC content per unit dry weight of soil. Data were collected for surficial soils and soil borings to a maximum depth of 18 ft. Data from soil boring samples include gravel- and cobble-sized particles. This measure of soil content, which may be more properly termed geologic materials, probably overestimates the actual amount of metal that is bioavailable, and therefore, overestimates the potential toxicity. Hence, only those data for surficial soils are presented and evaluated in the EE.

#### Methodology for Surface Waters

The concentration of COCs in surface waters was evaluated from data collected during the Surface Water Monitoring program at RFP. Data from surface water stations upgradient and downgradient from OU1 IHSS areas were examined for exceedance of RFP background concentrations and surface water quality standards. Data are presented for dissolved and total recoverable metals in surface water samples. The dissolved measure represents that fraction most available to aquatic biota and most appropriate for comparison with Colorado Water Quality Standards.

#### Methodology for Sediments

Data on contaminant distribution in sediments are also drawn from an ongoing monitoring program conducted at RFP. Sediment sampling stations have been established on Woman Creek and the South Interceptor Ditch directly south of OU1, but no data were available for these sites. Data were available, however, for sites upgradient and downgradient from OU1. Sediment sampling stations SED016 and SED017 are located on Woman Creek west (upgradient) of OU1 and correspond to surface water stations SW107 and SW041, respectively. Sediment stations SED018 and SED019 are located at ground-water seeps and correspond to surface water stations

SW080 and SW104, respectively. Station SED027 is located on Woman Creek just downstream from Pond C-1, and SED026 is located further downstream, just above Pond C-2. Stations SED028 and SED031 are both on the South Interceptor Ditch, downgradient from OU1 but upgradient from Pond C-2. Data for sediments were expressed as total COC content per unit dry weight.

#### E3.6.2 Uncertainty Analyses for Exposure Assessment

The major uncertainties and limitations in the exposure assessments include the following:

- Reliance on historical data
- Heterogeneity of sample data
- Extrapolation of potency estimates across routes of exposure
- Lack of or uncertainty in the data used to derive relative absorption factors
- Variation in plant uptake of metals, according to the specific element, soil characteristics, and plant species

The toxicity of most heavy metals is due primarily to the most reactive or organically transformed states. These forms of the metals are usually less stable and tend to form the more stable, less toxic forms under normal oxidizing or aerobic conditions. However, analysis of metals in soil, surface water and sediment included only total content of the metal. Therefore, measure of total metal content in environmental media will tend to overestimate the actual exposure of ecological receptors to the toxic forms of the metals.

#### E3.7 RISK CHARACTERIZATION

The purpose of the risk characterization is to integrate information from the Toxicity Assessment and Exposure Assessment sections to estimate the relative risk associated with exposure to COCs at OU1 (EPA 1989a, 1992a; Figure E3.1-1). Risk characterization actually began during the toxicity assessment phase with the iterative identification of COCs based on their expected concentrations, relative toxicity, and bioavailability. COCs were identified primarily on the basis

of comparison with RFP background concentrations. The risks associated with the COCs were further evaluated using the toxicologically-based reference values and the concentrations measured in environmental media at OU1.

#### E3.7.1 Use of Outputs from Toxicity and Exposure Assessment

The relative risks associated with exposures are evaluated by comparing the exposure concentration to the FRV. The relative magnitude of the exposure is assessed by dividing the FRV into the exposure concentration. The result is a ratio or quotient called the HQ. The HQ for a given COC  $i$  in an environmental medium  $s$  is calculated as:

$$HQ_{i,s} = \frac{[\text{exposure concentration}]_{i,s}}{FRV_{i,s}}$$

The greater the quotient, the greater the inferred risk of toxic effects (Figure E3.7-1). This approach is based on the HQ method developed for human health risk assessments (EPA 1989e) and adopted for environmental risk assessments (EPA 1989a, 1992a). As described in Section E3.5, the FRV was derived to approximate a concentration for a COC that is lower than a concentration expected to result in toxic effects (i.e., the NOAEL) to more sensitive species within an ecological community. Therefore, a HQ less than or equal to 1.0 would result from exposure concentrations at which no toxic effects would be expected, and HQ values greater than 1.0 represent corresponding increases to the levels of risk.

It should be stressed that FRVs are not intended to be action levels. Rather they are merely benchmark concentrations for evaluating potential hazards. The FRVs are conservative because they are based on sublethal effects to the most sensitive groups of species.

#### E3.7.2 Quantification of Risks from Individual and Multiple Chemicals

The level of risk represented by HQ values has been categorized as low, moderate, or high based on the safety factors used to derive the FRVs in Section E3.5 (Figure E3.7-1). For example, the

maximum ratio of the LOAEL to the NOAEL is about 3.5 in studies on vertebrates. In this EE, if the FRV approximates the NOAEL, a HQ value of less than or equal to 3.0 represents low risk; this value indicates an exposure approximating the concentration at or below the threshold for toxic effects. Continuing with this rationale, HQ values from 3.0 to 6.0 represent moderate risk; these values would indicate exposures exceeding the threshold for effects to sensitive species, but not exceeding the  $EC_{50}$ . Finally, a HQ value exceeding 6.0 corresponds to high risk; these exposures may affect greater than half of the sensitive populations and may result in toxic effects to more tolerant species.

Using this approach HQ values were calculated for screened COCs at each sample site in the OU1 study area; the mean HQ for the area was also calculated. HQ values from individual sites are used to assess risks attributable to isolated "hot spots." The mean HQ was used to evaluate the overall risk of areas within OU1 study area. For soils risks were evaluated for sites within or bordering an OU1 IHSS sites in OU1, but not associated with an IHSS, and background areas. Surface water and sediment sites were evaluated individually.

Risk due to exposure to multiple contaminants was evaluated using the sum of the HQ values for COCs in a given area. This approach is also adapted from human health risk assessments (EPA 1989e) and is referred to as the Hazard Index (HI). The value of the HI was evaluated using the same rating scheme as for HQ of individual COCs (Figure E3.7-1).

Although the HQ and HI approach as described here is generalized and semiquantitative, it allows standardized comparisons of relative risks associated with each of the chemical stressors at OU1. This approach also protects entire communities because higher HQ values indicate not only increased risks to the sensitive population upon which the FRV was based, but also potential adverse effects to more tolerant species. The higher the HQ, the more species likely to be affected. Risks to diverse taxonomic groups from direct exposure to soil and water are assessed separately since FRVs were established for organisms from different trophic levels of terrestrial and aquatic habitats.

Effects due to transfer of contaminants through food webs are not treated using the HQ approach (potential food web impacts were evaluated for ecological communities using procedures discussed in Section E3.8). However, whole body burdens of target analytes in plants and animals were measured for OU1 and reference area sites to determine gross concentrations of COCs. This measure does not assess the actual incorporation of target analytes into individual tissues, a measure needed to assess potential toxicity of accumulated contaminant loads, because nonavailable forms of COCs (e.g., minerals in soil ingested by organisms) were not quantified. Therefore, an evaluation of the background levels of COCs in featured species was used to assess whether OU1 sources had similar potential exposure to COCs. The COCs for the OU1 EE are predominately metals and radionuclides which, because of typically low bioavailability and low absorption, are not transferred efficiently between trophic levels.

#### E3.7.3 Uncertainty Analysis for Risk Characterization

The main sources of uncertainty in the risk characterization are those associated with the toxicity assessment and exposure assessment. Namely, the uncertainty associated with extrapolation of toxicity information from one site or study to another and with the sampling and analysis procedure for environmental samples. The sources of these uncertainties and the steps taken to account for them have been described in Sections E3.5 and E3.6.

The HQ method is commonly used to evaluate risks of noncarcinogenic toxins. However, the HQ method is conservative because exposure estimations do not include consideration of frequency, timing, or duration of exposure. The maximum exposure is assumed in calculating the HQ values at a given site. In addition, interpretation of the risk associated with HQ values can be subjective, especially when they approach the threshold value of an HQ value equal to 1.0. The use of the HQ method in this report was designed to account for these uncertainties where possible. Calculation of the HQ value from reference values that approximate COC concentrations results in low or no toxicological effects. The scheme used to rate risk using HQ values is based on empirically derived relationships between toxicological endpoints such as NOAEL and  $EC_{50}$  which may result in errors.

Other sources of uncertainty result from use of independent HQ values for each sample site. The exposure is qualitatively assessed, thereby highlighting "hot spots" that may be of limited areal extent and of limited actual risk. The mean HQ values for the areas within OU1 are used as an indication of the overall hazard of the site while the mean concentration for surficial soils is based on a minimum of 13 sample sites distributed around the IHSS and non-IHSS areas. These sites were randomly located within the sampled areas to provide representative data for soils. Therefore, the mean concentration of a COC is indicative of the potential exposure for a given area.

### E3.8 METHODS FOR ECOLOGICAL COMPARISONS

#### E3.8.1 Taxonomic Group and Trophic Level Comparisons

Species richness was calculated for terrestrial and aquatic taxonomic groups and trophic levels for comparisons between OU1 and Rock Creek, the reference area. Two computations were made for these comparisons—percentage and a chi-square statistic. The percentage was the amount each taxonomic group or trophic level in the food web contributed to total species richness for the area. The areas were then compared, looking for a difference between areas of over 30 percent. Thirty percent is within the range of natural variability. If a difference greater than 30 percent occurred, a more detailed evaluation, including life history requirements for species would be used to evaluate the variation in habitats at OU1 areas. This would entail making specific comparisons on the community level.

The chi-square test is called a row by column contingency table (Denenberg 1976). This test compares the frequencies of two or more observed distributions. In this case the distributions of taxonomic groups or trophic levels in OU1 compared to those in Rock Creek. The comparisons were made at the 95 percent confidence level.

### E3.8.2 Ecological Endpoints for Habitat Comparisons

Organisms were classified by trophic level (i.e., producers, herbivores) to examine potential risks not found through analysis of groups identified through traditional taxonomic classification. The total number of species in each trophic level was calculated and comparisons were made between numbers of species in each trophic level at the OU1 study area and the reference area.

Endpoints for plants, arthropods, and small mammals included total number of taxa and species richness by taxonomic group. These endpoints were calculated from field data and tabulated using the mean, standard deviation, and standard error from the results of the four sample sites at OU1. Habitat comparisons were made by using the four sites in the OU1 study area and four sites in the Rock Creek watershed for similar habitats. Methods for the specific groups (i.e., vegetation sampling methods or small mammal sampling methods) are contained in the Ecology Volume 5.0 SOP (DOE 1991c).

## E4.0 RESULTS

### E4.1 TOXICITY ASSESSMENT

The objectives of the toxicity assessment were to identify chemicals at OU1 that present significant threat to ecological receptors and to evaluate their potential toxicity. The result of this process is identification and ranking of potentially hazardous chemical stressors (COCs). The toxicity assessment also guides development of toxicologically based reference values—TRVs and FRVs—used as benchmark concentrations in evaluating the potential risk posed by the COCs. The methods for identifying COCs and developing the reference values that are reported below are described in Section E3.5.

#### E4.1.1 Conceptual Model

A conceptual model was developed in Section E3.2 to identify the exposure pathways and exposure points to be evaluated in this EE. The major exposure pathways identified from this process were direct exposure to soil and surface water. Exposure through trophic interactions were identified as minor components since the COCs are primarily heavy metals.

The exposure points for soil and surface water evaluated in the risk characterization include: soils in the OU1 IHSSs and downgradient from the IHSSs; and surface water in the South Interceptor Ditch, Woman Creek, and detention Ponds C-1 and C-2. Soils in the IHSSs are the primary source areas for COCs and the location of highest potential exposure (Figure E3.2-2). Secondary release from the IHSSs may have occurred by surface runoff, possibly contaminating soils in downgradient areas. Surface runoff is also the primary mechanism by which surface water in Woman Creek and the South Interceptor Ditch may become affected.

#### E4.1.2 Identification of Contaminants of Concern

An initial list of COCs was established during the planning stages of the EE (Table E3.5-1). The COCs included heavy metals, cyanide, and radionuclides that had been detected at concentrations above background in soils, surface water, or sediments at OU1. The results of Phase III investigations were then used to screen the initial list as described in Section E3.5.3. The result



of this second level screening was identification of those COCs for which risks were characterized. The second stage screening process was based primarily on exceedance of RFP background concentrations, and secondarily on relative toxicity and bioavailability. The screening process identified chromium, lead, and zinc for soils; chromium, lead, and mercury for surface water; and no COCs for sediments at OU1 (Figures E4.1-1, E4.1-2, and E4.1-3, respectively).

#### E4.1.3 Development of Reference Concentrations

Benchmark reference values, TRVs and FRVs, were developed for each of the initial COCs identified during the planning stages of the EE. The TRVs and the references on which they were based are presented in Table E4.1-1. The FRVs derived from the TRVs and RFP background concentrations are presented for estimating potential exposure to surficial soils (Table E4.1-2) and surface water (Table E4.1-3).

### **E4.2 EXPOSURE ASSESSMENT**

The exposure assessment evaluates actual or predicted exposure concentrations, pathways, and ecological receptors to contamination resulting from IHSSs within the OU1. Potential exposures have been estimated based on concentrations of contaminants measured in soils, surface water, sediments, and biota collected in the study area. Much of the data for this assessment is found in Section 4.0 of the current RFI/RI. The potential hazard due to these exposures has been evaluated by comparing the estimated exposures to the reference values presented in Section E4.1. Results presented in this section are used in the risk characterization (Section E4.3).

The data used in the exposure assessment were drawn from the following sources:

- Operable Unit 1 Phase III RFI/RI Workplan (DOE 1991a)
- Phase III Site Characterization (Section 4.0 of the main RFI/RI report)
- Phase III Ecological Characterization (Section E2.0)
- Results of tissue analysis

Data presented in Section 4.0 of the current RFI/RI report are not re-presented in this Appendix. Rather, data are summarized for those sites with samples exceeding the RFP background, regulatory standards, or reference values. Surface water and sediment data are presented for the period January 1990 to December 1991.

#### E4.2.1 Fate and Transport of Selected Metals at OU1

The metals chromium, lead, mercury, and zinc have been identified, following the screening process shown in Figure E3.5-1, as contaminants with potential ecological impacts. The limited occurrence of these metals at levels above RFP background at OU1 have not been attributed to any releases from RFP. Neither are they identified as COCs in the PHE. It was deemed prudent to evaluate the potential risk that these metals pose to the biota at OU1 due to the known toxicity of these elements to various ecological receptors.

The presence of trace elements in soils and surface waters at concentrations greater than background is a localized and relatively insignificant occurrence at OU1. Background values for these elements vary widely at RFP, due in part to the existence to the wide variety of mineral types in the Rocky Flats alluvium and associated unconsolidated deposits. The limited occurrence of chromium, lead, mercury, and zinc at levels slightly above background, and the absence of these metals as contaminants of concern in the PHE, strongly suggest that these metals are not chronic contaminants, but more likely exist as natural outliers relative to background levels.

This section examines the mobility and availability of these metals to the environment at OU1 given observed site conditions. Natural sources of these elements are known to occur in the igneous rocks that supplied material to the Rocky Flats alluvium and are the probable source of most of these elements at RFP and OU1. Moreover, several of these metals are important trace nutrients for biological organisms.

## Chromium

Chromium has been identified as a contaminant with potential ecological impact at OU1 in surface waters and surface soils. Two surface soil locations at OU1 occur at concentrations greater than twofold background levels. These occur at RAO31 (107 mg/kg) and RAO18 (64.2 mg/kg). The background level for chromium at Rock Creek (background location for OU1 surface soils) is 18.3 mg/kg (Table E4.1-2). Chromium was identified as a contaminant in surface water because concentrations exceeded sitewide RFP background by more than twofold at one of six surface water stations along the South Interceptor Ditch at OU1. Chromium was detected at 63 µg/l at surface water station SW-070. The background for chromium in surface water at RFP is 10 µg/l (Table E4.1-3).

Geochemistry of Chromium – Chromium concentrations at OU1 are a measure of total chromium; thus, the fraction existing in hexavalent form is unknown. This determination can only be made by specifically testing for hexavalent chromium. However, given the geochemistry of chromium and the site conditions at OU1, the probable form of chromium in OU1 soils and surface water can be estimated.

Chromium is a transition metal having two stable oxidation states in natural environments, occurring as trivalent and hexavalent chromium. Hexavalent chromium is toxic to plants and animals, whereas trivalent chromium is considered to be less toxic. In rock and soil minerals the predominant form present is trivalent chromium. Hexavalent chromium is highly instable in natural environments and is readily reduced to trivalent chromium, especially in soil environments. Hexavalent chromium is only stable in highly oxidizing environments in the absence of other more oxidizable compounds, such as ferrous iron or organic carbon.

Hexavalent chromium occurs at very low levels in nature due primarily to kinetic limitations on the oxidation of trivalent to hexavalent chromium. Reduction of hexavalent chromium by photoreduction in surface waters is also known to occur. Low levels of trivalent chromium in aqueous systems is due to the formation of chromium (III) hydroxide, which is of very low

solubility. In contrast, there are few hexavalent solid phases to regulate the solubility of dissolved hexavalent chromium. As a result, the dissolved hexavalent chromium ion is highly mobile in aqueous systems.

Chromium in surface water is likely to be trivalent chromium due to kinetic constraints on hexavalent formation in natural systems. If trivalent chromium is oxidized, the hexavalent anion will be very mobile and migrate downward with infiltrating waters to deeper soil depths. However, in the organic-rich near surface soil environment, the preferential oxidation of organic matter is likely to prevent the oxidation of trivalent chromium. At the alkaline pH values in OU1 soils (9.7) the solid chromium hydroxide is stable.

Given the existing conditions at OU1 it is not likely that hexavalent chromium will be the dominant form of chromium. When and if hexavalent chromium is produced in the near surface environment, the high solubility and mobility of this species will remove a significant portion of it to deeper levels where it will encounter lower pH conditions and become susceptible to reduction.

#### Lead

Lead was identified as a contaminant of potential ecological concern in soil and surface water. Lead exceeded background in surface soils at surface soil location RA018, at a concentration of 228 mg/kg. Background for lead is 40 mg/kg (Table E4.1-2). Lead levels exceeded background at one location in the South Interceptor Ditch at OU1. Location SW064 detected lead at 38.4 µg/l. RFP background levels for dissolved lead in surface water is 4 µg/l (Table E4.1-3).

Geochemistry of Lead – Natural sources of lead include the minerals galena, anglesite, cerussite, and lead (II) hydroxide, all of which are known to occur in igneous rocks west of RFP. Another potential source of lead may be from scrap metal and drums stored at IHSS 119.1 where surface soil sampling location RA018 is located.

Lead is stable in one oxidation state in nature, as lead II. Dissolved forms of lead include the free ion, hydroxide complexes, carbonate, and sulfate ion pairs. The mobility of dissolved lead in natural environments is typically low because it combines readily with common anionic ligands (carbonates, sulfates, and hydroxides) and precipitating solids of low solubility. Lead solubility is also reduced by adsorption by organic matter and inorganic mineral surfaces (clays, sesquioxides), and is co-precipitated with manganese oxide. Lead also occurs in calcium carbonates, and is fixated in carbonate-rich soils. In contrast, complexation with dissolved organic carbon can increase lead solubility as an organo-metallic complex.

The solubility of lead in alkaline pH soils is extremely low. As a result, lead mobility and availability at OU1 is expected to be low. In addition, the very limited aerial extent of lead in soils at OU1 indicates that the availability to biota will be minimal.

### Mercury

Mercury was detected above background in surface water at RFP at one location (SW035) at a concentration 1.0 µg/l. Background mercury in surface water is 0.3 µg/l. Mercury use as a pesticide/herbicide is documented at RFP and its use may be a source of mercury at OU1 (Table E4.1-3).

Geochemistry of Mercury – Mercury exists in two oxidation states, as mercury I and II. Aqueous mercury as neutral pH include the neutral species mercury and mercuric chloride at acidic pH, and mercury hydroxide at alkaline pH. Methyl mercury may exist in organic-rich reducing conditions, but is unstable in oxidizing environments. Dissolved mercury compounds are regulated by solid mercurial chlorides and mercury oxide.

Sources of mercury include various mercury minerals, the most abundant being cinnabar. Cinnabar can be formed in reducing environments such as lake and streambed sediments, where sulfide ion and methanogenic bacteria is present. Mercury is absorbed by soils and plant matter. Mercury forms a wide variety of inorganic and organic compounds. Mercury tends to

concentrate in clay-rich sediments and soils, especially those rich in iron and manganese, where adsorption or co-precipitation processes are operating.

Mercury concentrations are typically low in most waters because mercury readily volatilizes and is lost from surface water to air, and mercury has a large organic carbon partition coefficient and partitions to organic matter readily. This removes it from the dissolved phase and lowers its mobility.

### Zinc

Zinc was identified as a contaminant of potential ecological impact in soils at OU1. Zinc was detected in surface soils at two locations (RA014 and RA031) at concentrations of 182 mg/kg and 165 mg/kg, respectively. RFP background for zinc in soils is 71 mg/kg (Table E4.1-2).

Geochemistry of Zinc – Zinc has only one important oxidation state in natural systems as a divalent cation. Zinc is relatively soluble in most natural waters, occurring paired with hydroxides, carbonates, and as the free ion. Solid phase solubility controls include the precipitation of zinc hydroxide at alkaline pH. Zinc mobility can be lowered by adsorption in clay-rich environments and by co-precipitation in alkaline.

Zinc mobility in surface soils is expected to be moderate to low given the alkaline nature and high clay content of OU1 materials.

### E4.2.2 Soils

The primary receptors exposed to potentially contaminated soils at OU1 are terrestrial vegetation and invertebrates. The primary exposure mechanism for vegetation is uptake by roots of COCs in solution. Invertebrates may take up COCs in soils by transcuticular uptake or ingestion of organic material during feeding. Minor exposure pathways include incidental ingestion of soils by grazing or burrowing animals and respiratory uptake of re-suspended particles. Data from analysis of soil contamination are detailed in Section 4.2 of the RFI/RI report.

Soils in IHSSs constitute the secondary source areas as the original contaminant sources have been removed. These soils represent the major source for direct exposure to contaminated media and potentially for further transport of contaminants away from the original source areas by eolian or erosional transport. Soils within OU1 but remote from the IHSS areas are likely to be impacted by eolian and erosional transport of contaminated soil particles away from the IHSS source areas. These soils are considered separately in the risk assessment.

Those analytes detected in OU1 soils at concentrations exceeding RFP background are listed in Figures E4.2-1 and E4.2-2. Most of these analytes were detected at concentrations only marginally (greater than twofold) above background. As noted previously, the total amount metals and radionuclides were measured in soil samples. The contribution of reactive or organically transformed states, expected to be minor quantities, to the total concentration of COCs was not measured. Since the reactive forms are the most toxic forms of many heavy metals, this measure of metal concentration in soils overestimates exposure to the toxic forms of the metals.

#### E4.2.3 Surface Water

Surface water quality is linked to the chemistry of the soils in a drainage basin as soils transported to surface water become sediments and can impact water quality. Many aquatic organisms are highly sensitive to contaminants carried with the soil particles. The major pathways for exposure of aquatic organisms are direct contact with dissolved contaminants in water and subsequent absorption and indirect exposure to particles adhering to external body surfaces.

Exposure of ecological receptors to COCs in surface water was assessed using monthly data from the surface water monitoring program at RFP. Data from January 1990 to December 1991 were examined for sites on Woman Creek and the South Interceptor Ditch. Data for total recoverable and dissolved metals and radionuclides was screened for exceedance of RFP background and Colorado Water Quality Standards for protection of aquatic life. For regulatory purposes,

dissolved concentrations are the appropriate comparison because the Colorado standards are intended for use with this measure. EPA's Ambient Water Quality Criteria were developed based on the measures of metal content using an acid-soluble extraction technique (the strong acid dissolves the most reactive elements). Since EPA has not approved a method for acid-soluble extraction, it suggests that the total recoverable method be used for comparison to suggested criteria. Colorado standards for chemical content of water used for watering of livestock were used to assess potential risk to wildlife drinking from OU1 surface waters. These standards are based on total recoverable metals and radionuclides. Data for dissolved concentrations are summarized in Figure E4.2-3. As noted previously, Woman Creek's intermittent flow and the low average discharge volume may qualify for classification as a Class 2 stream under Colorado water quality statutes. Highly variable chemical content may be related to the small drainage area, intermittent flows, and proximity to ground water sources.

#### Woman Creek Sites

Water from several stations along Woman Creek contained dissolved metal concentrations that exceed RFP background concentrations (Figure E4.2-3). However, few of the sites sampled exceeded Colorado standards (Figure E4.2-4). Exceedance of background or regulatory standards was infrequent at most sites suggesting correlation with surface runoff events. Surface water sampling sites along the South Interceptor Ditch exhibited only sporadic exceedance of RFP background and state standards when total concentrations were assessed (Figure E4.2-4). SW035 contained the highest metal concentrations and most frequent exceedances of RFP background and Colorado standards. SW035 is just west of the IHSSs contained in OU1 and is probably upgradient from these sites. Therefore, the source of the high metal values at this station may not be related to activities at OU1, but to upgradient sites. Based on the criteria described for identification of COCs, the risk due to mercury, chromium, and lead is assessed in Section E4.3.

#### Radionuclides in Surface Water

Radionuclides in surface water samples did not exceed RFP background or FRV values, except infrequently for uranium isotopes collected from stations along the South Interceptor Ditch during



the period July through September 1991. The maximum concentration of uranium was 5.2 picoCuries per liter (pCi/l) (uranium-233,-234 dissolved), which occurred at SW046. As noted in Section 4.0 of the RFI/RI report, radionuclide levels at OU1 probably resulted from windborne dispersal of contaminated dust from the 903 Pad area (OU2 study area).

#### Drinking Water Sources for Wildlife

At no sites along Woman Creek or the South Interceptor Ditch did COC concentrations exceed the FRV for drinking water use by wildlife. Station SW080, a ground-water seep on the opposite side of the drainage, produced a single sample that exceeded the FRV for manganese. The sample appears to represent an isolated case that could have resulted from the difficulty in sampling surface water at seeps.

#### Aquatic Toxicity Screening

For screening purposes, acute aquatic toxicity bioassays were conducted on samples collected from Woman Creek according to methods presented in Section E3.3.2. Results of the toxicity test screens are summarized in Figure E4.2-5. Significant toxicity to water fleas was detected at nearly all stream sites upstream of Pond C-2, including sites upgradient of OU1 and outside of any apparent impact of the industrial area of RFP. No significant toxicity to water fleas was detected at Ponds C-1 and C-2, or at stream sites downgradient of C-2. The only significant toxicity to fathead minnows was detected at SW104, a ground-water seep on the opposite side of the Woman Creek drainage from the RFP industrial area.

As might be expected for the headwater area of a stream, a relatively wide range of hardness and alkalinity values (range 86 to 220 mg/l and 88 to 290 mg/l, respectively) were encountered. Both parameters increased with distance downstream from the headwaters. The pH measured during the tests of all samples ranged from 7.0 to 8.7. Detailed water chemistry for sites on Woman Creek is presented in Section 4.0 of the RFI/RI report.

#### E4.2.4 Sediments

Sediment sampling stations have been established on Woman Creek and the South Interceptor Ditch directly south of OU1, but no data were available within the OU1 study area. Data were available, however, for sites upgradient and downgradient from OU1. Sediment sampling stations SED016 and SED017 are located on Woman Creek west (upgradient) of OU1 and correspond to surface water stations SW107 and SW041, respectively. Sediment stations SED018 and SED019 are located at ground-water seeps and correspond to surface water stations SW080 and SW104, respectively. Station SED027 is located on Woman Creek just downstream from Pond C-1, and SED026 is located further downstream, just above Pond C-2. Stations SED028 and SED031 are both on the South Interceptor Ditch, downgradient from OU1 but upgradient from Pond C-2. Data on analysis of sediments collected from the above stations is summarized in Figure E4.2-6.

Aluminum, iron, and arsenic were the only COCs present above background concentrations in sediment samples at stations listed above. Only aluminum exceeded background values within the OU1 study area (Figure E4.2-6).

#### E4.2.5 Biota

Vegetation, terrestrial insects, small mammals, and fish were analyzed for whole body burdens of COCs as described in Section E3.3.3. The purpose of this analysis was to identify increased uptake of contaminants due to elevated levels in environmental media at OU1. Most of the target analytes were metals or radionuclides and do not tend to bioaccumulate due to transfers among trophic components. However, many metals tend to bioconcentrate in many aquatic organisms. Species collected for tissue samples from the OU1 study area and the Rock Creek reference sites are listed in Table E4.2-1. Samples collected from reference areas in the Rock Creek drainage were used to characterize background concentrations in biological tissues (Table E4.2-2). Samples from the OU1 study area that exceeded the background concentration for a given metal are given in Tables E4.2-3 through E4.2-6.

### E4.3 ECOLOGICAL RISK CHARACTERIZATION

The risk characterization focuses on evaluation of the potential impacts of the COCs identified as most hazardous based on their toxicity and the concentrations at which they were detected at in environmental media at OU1. The COCs with potential risks were identified for soil, surface water and sediments, the main exposure points named in the conceptual model (Figures E4.1-1, E4.1-2, and E4.1-3). COCs identified for soils are chromium, lead, and zinc; for surface water, chromium, lead, and mercury; and none for sediments (Table E4.3-1).

The remaining risk characterization focuses on toxicological risks from exposure to each of these chemicals and the aggregate risk of simultaneous exposure to these COCs.

#### E4.3.1 Risks from Individual COCs Chromium

Within OU1, chromium was detected in surficial soils from several sample sites at concentrations marginally above RFP background (less than twofold) (Figure E4.2-2). Concentrations of this magnitude are not outside the range of chromium concentrations detected in background samples from the Rock Creek area (Table E4.3-2). However, chromium was detected at concentrations greater than twofold background from two sites in IHSSs 119.1 and 119.2 (Figure E4.2-1 and Table E4.3-2). Chromium concentrations in these two samples were significantly greater than other samples collected from within the same IHSSs, as well as from other IHSSs within the OU1 study area, suggesting a highly restricted distribution of elevated chromium in surface soils. When these two samples are excluded, the mean concentration of chromium in soils from the IHSS areas is  $16.2 \pm 6.15$  mg/kg, which is not different from RFP background areas or from soils in non-IHSS areas within OU1 (Table E4.3-2).

When risks from chromium were assessed using the HQ method as described in Section E3.7, the risk level associated with the highest chromium concentrations in IHSSs 119.1 and 119.2 was moderate (Table E4.3-2). This assessment is based on HQ values of 3.5 and 4.4 calculated for exposure of vegetation and soil invertebrates to the two highest soil concentrations of chromium. Risks from exposure to chromium for all other sites in the OU1 area are low with HQ values of

1.5 or less. The overall risk to ecological receptors from exposure to chromium in soils in the OU1 area is low. This judgement is based on the highly restricted distribution of high chromium concentrations and a mean HQ value of 1.3 for the IHSS areas, and 0.83 for non-IHSS areas within OU1.

Chromium was also detected in surface water at concentrations above RFP background (Figure E4.2-3). However, only a single sample, collected from SW070, was more than twofold above the background concentration for dissolved chromium. No samples exceeded the (chronic) Colorado Surface Water Quality Standard for total chromium of 170 µg/l. As noted previously, the concentrations of total chromium present as trivalent chromium and the less stable hexavalent chromium are not known. However, it is probable that greater than 99 percent of the total chromium is present as the less toxic trivalent chromium. The concentration of chromium in the sample was 63 µg/l, resulting in a HQ value of 0.36, corresponding to low risk to ecological receptors. All other HQ values for the site are 0.1 or less. These HQ values are calculated using the FRV of 172 µg/l, which is equal to the water quality standard.

Station SW070 is located in the South Interceptor Ditch, downgradient from IHSSs 119.1 and 119.2 suggesting a pathway from the OU1 source areas (Figure E4.2-3). However, the South Interceptor Ditch empties to Pond C-2, the terminal pond in the South Interceptor Ditch flow path before the water is treated and pumped to Pond A-5, which is a National Pollutant Discharge Elimination System (NPDES) permitted discharge point.

Pond C-2 is used by wildlife and contains a limited ichthyofauna in the OU1 study area. However, surface water and sediments of Pond C-2 do not contain elevated levels of chromium. Thus, the restricted distribution of chromium at OU1, and lack of downgradient contamination indicate low risk to ecological receptors from exposure, and little or no risk of off-site movement.

### Zinc

The distribution of zinc in soils at OU1 was similar to that of chromium with average zinc concentrations marginally above RFP background and three of 28 sample sites exceeding twofold RFP background concentrations (Table E4.3-3). The highest concentrations of zinc were detected in IHSS 119.2 and the area of the Building 881. As with chromium, potential zinc contamination was restricted, as the mean concentration of IHSS areas, excluding the three highest concentrations, was  $70.73 \pm 15.2$  mg/kg, which is not different from non-IHSS areas within OU1 ( $60.19 \pm 4.53$  mg/kg), or from background areas ( $61.98 \pm 12.74$  mg/kg).

The three highest zinc concentrations in soils resulted in HQ values of 2.56, 2.32, and 1.83 for exposure of vegetation and 0.18, 0.17, and 0.13 for exposure of soil invertebrates (Table E4.3-3). Each of these values correspond to low environmental risk. The overall HQ values for vegetation and soil invertebrates were 1.22 and 0.09 in the IHSS area, and 0.85 and 0.06 in the non-IHSS areas, also corresponding to low environmental risks.

Contamination of surface water or sediment by transport of zinc from IHSS areas was not apparent as zinc was not elevated in either of these two media. Zinc is known to accumulate in some plant species, but levels were elevated only slightly in vegetation from the OU1 areas (Table E4.2-2) and may be attributed to adherence of soil particles on the surface of plants.

### Lead

Lead was detected at concentrations above background in soils at four sites in and around IHSSs 119.1 and 119.2 (Table E4.3-4). Lead concentrations at three of the sites were about 10 percent above background. However, a single sample from IHSS 119.1 contained lead at a concentration of 228 mg/kg, more than four times the RFP background concentration. A second sample from the same site within IHSS 119.1 contained lead at 78 mg/kg. These data suggest that lead contamination in soils at OU1 is highly localized in areas within IHSS 119.1. As with chromium and zinc, when the isolated high concentration is excluded, the mean lead concentration in OU1 soils is well within the range of background concentrations.

When based on the single high concentration from the single sample, exposure to lead within IHSS 119.1 could result in moderate risk to vegetation and soil invertebrates. HQ values were 4.6 and 5.7, respectively. All other sites within the IHSS and non-IHSS areas of OU1 are characterized by low risk to ecological receptors with HQ values less than or equal to 1.0 for all sites. The overall risk associated with lead in soils in OU1 is low because of the restricted distribution within IHSS 199.1 and because of possible sampling bias in initial samples.

Although lead had restricted distribution and infrequent occurrence above background concentrations at OU1, lead was also detected at elevated concentrations in surface water of Woman Creek drainage and the South Interceptor Ditch. Marginal elevation in lead content of water samples were detected in Woman Creek at areas upgradient and downgradient of the OU1 area (Figure E4.2-3). The highest lead concentrations in Woman Creek water samples was 13.2 µg/l at SW034. However, this sampling station is located on a branch of Woman Creek which drains areas south of the plant site, and upgradient of the OU1. Lead was also elevated in some sediment samples from these upper drainages as well as sediment stations downgradient of Pond C-1 (Figure E4.2-6). Therefore, elevated lead levels detected in reaches of Woman Creek downgradient of OU1 may be due to sources upgradient of OU1, and therefore not strictly attributable to sources within OU1. The maximum concentration of lead in surface water, 38.4 µg/l (dissolved), was detected at SW064 on the South Interceptor Ditch (Figure E4.2-3). However, lead levels of this magnitude were detected on only one sampling date. Samples from the same station at other times contained lead concentrations only slightly above RFP background. Sediments of Woman Creek and the South Interceptor Ditch downgradient of IHSS 199.2 did not contain elevated lead concentrations.

The overall risk to ecological receptors due to lead exposure at and around OU1 is considered low. This judgment is based on the generally low level of lead in soils, surface water, and sediments, and the restricted distribution of areas of higher lead concentrations. The HQ value associated with the highest lead concentrations in surface water is 9.6, corresponding to high environmental risk. The overall risk to aquatic organisms from exposure to lead is considered

low because lead concentrations are generally low, with infrequent high concentrations. Furthermore, the South Interceptor Ditch was built to intercept contaminants from OU1 preventing contamination of Woman Creek. Flow from the South Interceptor Ditch is captured in Pond C-2 for treatment.

Biological tissue data from the OU1 area indicate minimal uptake of lead. Lead content of small mammal and vegetation tissue samples collected from sites throughout the OU1 area including IHSS and non-IHSS areas was slightly higher than background samples (Rock Creek watershed). Lead concentrations were also slightly higher than background for fish collected from Woman Creek above Pond C-2. However, most samples which exceeded the background concentrations were bottom-feeding fish such as white-suckers and creek chubs. Since fish were analyzed with GI tracts intact, the apparent elevated lead levels could be due to sediment particles contained in the gut at the time of analysis.

Lead seems to be the most ubiquitous of the potential contaminants at OU1. It is elevated in soils and surface water, and marginally elevated in biological tissues. However, hazardous concentrations of lead are highly restricted to IHSS 119.1 and surrounding areas. By comparison, lead concentrations in soils near lead mines and other areas of documented effects of lead contamination range from 1,500 to greater than 5,000 mg/kg. EPA currently recommends that soils at lead contaminated sites be remediated to levels of between 500 and 1,000 mg/kg. The maximum concentration of lead in soils at OU1 was 228 mg/kg. A decision to remediate the site based on the risk from lead concentrations alone is not justified at this time. A focused study of lead contamination first must be evaluated to determine the quantity of soil with elevated lead levels at OU1.

### Mercury

Mercury concentrations did not exceed background in OU1 soils (Figure E4.2-1), nor in surface water or sediments in Woman Creek downgradient from OU1 IHSSs (Figures E4.2-3). Mercury concentration in water samples did exceed background in surface water at SW080 and SW107,

two sites on branches of Woman Creek that drain areas south of the RFP industrial area. However, these areas are outside potential impact from OU1 areas and the concentrations exceeded background by less than 30 percent. Mercury concentrations did exceed background in surface water samples collected from SW035 and SW031. Both of these stations are located on the South Interceptor Ditch near the 881 Hillside area. However, SW035 is west of the IHSSs in this area and so is likely upgradient of OU1 sources. The highest concentration of mercury in surface water, 1.0 µg/l, was detected at SW035, with the concentration at SW031 measured at 0.56 µg/l in one sample. Since there is no probable source within OU1, it is likely that the source of mercury in the South Interceptor Ditch is upgradient of OU1 areas, possibly one of the OU5 IHSSs.

The risk associated with a mercury concentration of 1.0 µg/l is judged as moderate, with a HQ value of 3.3, based on RFP background concentration of 0.3 µg/l. The risk associated with mercury concentrations at the other stations is low with HQ values of 1.9 or less. The acute and chronic Colorado standards for mercury are 2.4 and 0.1 µg/l, respectively. The overall risk to ecological receptors from mercury exposure at OU1 is judged to be low. This is based on the infrequent exceedances of background levels in surface water and the lack of evidence for contamination in other media, including biological tissues collected from the site.

#### E4.3.2 Combined Risks from Exposure Points

The sum of the mean HQ values for exposure of vegetation and soil invertebrates was used to assess the overall risk attributed to soils in OU1 IHSS and non-IHSS areas. The HI values for vegetation in IHSS areas, non-IHSS areas, and background areas were 3.4, 2.4, and 2.5, respectively; the values for the same components for soil invertebrates were 2.5, 1.8, and 1.9, respectively. This suggests that vegetation growing in the IHSS areas may be at moderate risk from exposure, and that vegetation in non-IHSS areas, and soil invertebrates in all areas are at low risk. For both taxa, the HI for non-IHSS areas in OU1 was not different from that calculated for the background areas in the Rock Creek drainage.



As noted in the discussion of risks from individual COCs (Section E4.3.1), high concentrations of each COC is an overestimate of the actual exposure at OU1. Moreover, elevated concentrations of each of the COCs is restricted to a single IHSS or surface water sampling station within OU1. The mean HI probably overestimates the actual risk that should be attributed to an individual COC and inflates estimates of the aggregate risk to ecological receptors from the site as reflected by the HI. The highest concentration of chromium, lead, and zinc were found in the areas of IHSSs 119.1 and 119.2. Even within these IHSSs concentrations of these chemicals were highly localized (Figure E4.2-1). Based on this restricted distribution of contaminants in soils at OU1, the overall risk to ecological receptors from exposure to multiple contaminants in soils at OU1 is assessed as low.

The risks to ecological receptors from exposure to multiple contaminants in aquatic habitat is also low. Mercury and lead were the only analytes to exceed background by more than twofold at any surface water or sediment station downgradient of OU1 source areas. Their distribution was restricted to the South Interceptor Ditch which was constructed and is maintained as a wastewater collection system. The South Interceptor Ditch provides limited and low quality aquatic habitat because of frequent absence of surface water. The South Interceptor Ditch drains to Pond C-2, the terminal pond in its path. Data from the surface water monitoring program did not indicate concentration of the COCs exceeded RFP background. It should be noted that data on sediments were not available for Pond C-1 or C-2 and therefore information on contamination of sediments in the ponds is lacking.

Surface water quality in Woman Creek is generally good. The concentrations of some analytes did exceed RFP background and Colorado Water Quality Standards at stations downgradient of OU1 source areas (Figure E4.2-4). However, elevated concentrations were also detected at stations upgradient of OU1, and may be the source of the elevated levels at downgradient stations. At any rate the risk to ecological receptors from elevated concentrations of these analytes in Woman Creek is low because of the low frequency of occurrence and restricted locations of samples with elevated values in OU1.

## E4.4 ECOLOGICAL COMPARISONS AT OU1

### E4.4.1 Taxonomic Group Comparisons

Terrestrial and aquatic ecosystems were compared using species richness at both a taxonomic level and a trophic level. These comparisons were made to assess potential ecological effects that suspected OU1 contaminants may have on the terrestrial and aquatic ecosystems. As stated in Section E3.8, comparisons of percentage differences were made between taxonomic groups at OU1 and Rock Creek watershed.

Table E4.4-1 presents results of the taxonomic group comparisons using percentages. The terrestrial ecosystem revealed no percentage difference (i.e., the difference between the percentage of OU1 mammalian species and reference mammalian species) over the 30 percent threshold. In fact, there were no percentage differences greater than 3 percent, indicating similar species richness in the two areas.

The aquatic ecosystem also showed no percentage differences over 30 percent. There were differences of approximately 10 percent between OU1 streams and Rock Creek area plankton and benthic macroinvertebrate richness. Rock Creek had 11 percent more plankton species than OU1 streams. Benthic macroinvertebrate richness was greater by 9 percent in the OU1 area. Plankton species are characterized by dynamic population cycles of short duration. During a population decline, many species that are present only in low numbers may be missed during sampling. Benthic macroinvertebrates, unlike plankton, do not go through rapid population fluctuations, and individuals can live up to 3 years. The comparison of species richness for benthic macroinvertebrates at OU1 streams and Rock Creek indicates similar levels of ecosystem health.

Chi-square calculations (Table E4.4-2) showed no significant difference between areas for either terrestrial or aquatic taxonomic richness. In fact, the relatively small value of chi-square indicated a good similarity between OU1 and Rock Creek watershed for both terrestrial and aquatic ecosystem elements that were evaluated.

#### E4.4.2 Food Web Comparisons

The species used in the taxonomic level comparison were organized by trophic levels for a food web comparison and are presented in Table E4.4-3. Terrestrial arthropods were compared separately due to a less detailed identification endpoint due to study objectives. As in the taxonomic comparisons, trophic comparisons revealed much similarity between the OU1 and the Rock Creek ecosystems. The maximum percentage difference for any trophic level between these two areas was approximately 4 percent. Comparisons of terrestrial arthropods showed differences of only 6 percent or less between the two areas.

Trophic level comparisons for aquatic primary producers and omnivores showed differences up to 10 percent between OU1 streams and Rock Creek. The comparison of aquatic primary producers does not differ from the plankton comparison in Table E4.4-3. Comparison of species richness for omnivores at the two areas revealed greater species richness at OU1, reflecting good ecosystem health in the OU1 study area.

Chi-square calculations for terrestrial trophic levels (Table E4.4-4) revealed no significant difference between areas, again indicating much similarity between areas. Aquatic trophic levels, however, did show a significant difference between areas. This is due to the total number of aquatic taxa being higher in OU1 streams than in Rock Creek, although primary producers (algae) were higher in Rock Creek. Rock Creek tends to be more intermittent than Woman Creek and, therefore, is a more harsh environment for aquatic organisms. This fact gives an explanation for a diminished number of taxa when compared to the OU1 area in Woman Creek watershed.

Results for food web comparisons did not indicate a stressed ecosystem, nor did they reveal ecological problem areas. These results, along with those of tissue sample analysis in Section E4.2.4, should be viewed as further weight-of-evidence indicating an ecologically healthy state at the OU1 study area.

#### E4.4.3 Ecological Habitat Comparisons

Plant and animal endpoints were compared by habitat type using the mean, standard deviation of a population, and a standard error (Table E4.4-5). While reviewing this endpoint data, three general sources of variability became apparent. The first, dealing with the xeric habitat sites, is that only one sample site for OU1 could be established within the ecological study area boundary (Figure E2.2-1). Four sites were established in the Rock Creek reference area, this being the normal protocol. The low number of replicates available in OU1 causes variability in the data in that the reference results were derived from four times the number of observations. In addition, the soil in the single OU1 xeric site was extremely compacted which physically inhibits plant growth and small mammal burrowing. Second, the marsh habitats in OU1 and Rock Creek comprise different wetland plant types. The OU1 marsh habitats are the result of construction activities that created the South Interceptor Ditch and Ponds C-1 and C-2. These areas contain mainly cattails and open water. The vegetative structure differs from marsh habitat found in Rock Creek, which are mostly the result of ground water seeps and are more suitable habitat for rushes and wet grass species. Although both areas are hydric in nature, these site differences explain the considerable variation of species richness and productivity of both plants and animals. Finally, results for ecological comparisons do not indicate a difference that may be the result of contaminants.

Each habitat type in Table E4.4-5 was reviewed for the same endpoints along the moisture gradient, dry to wet habitats. Endpoints were compared within habitats by the general groupings of plants, terrestrial arthropods, and small mammals.

Xeric grassland habitat endpoint comparisons varied greatly between OU1 and Rock Creek (Table E4.4-5). Tree and shrub density and plant species richness were inhibited by soil compaction in OU1, although standing crop measurements indicated higher production in the OU1 study site. The number of individuals of arthropods in OU1 study sites was considerably smaller than at Rock Creek sampling sites. Small mammal trapping efforts in the OU1 xeric site was quite unsuccessful in capturing any small mammals during both spring and fall. No small

mammals were captured in spring and only one plains harvest mouse was captured in fall. Six individuals of one species (deer mice) were captured in spring and thirteen individuals of two species were captured in fall in the Rock Creek reference area. The soil compaction in the xeric site at OU1 physically limits small mammal burrowing. This fact plus the difference in number of trap nights (100 for one OU1 site versus 400 for four Rock Creek sites) provides an explanation for the large differences in small mammal endpoints (Table E4.4-5).

All endpoint means from mesic grassland habitat were within data standard deviations between the two areas. For example, the endpoints for OU1 and Rock Creek sample sites were quite comparable in that no large differences were observed. Tree and shrub density in OU1 was zero, but this mean value falls within the standard deviation of the Rock Creek reference area and, therefore, is not a significant difference. Mesic plant species richness and standing crop were higher in OU1 sites than at Rock Creek sites. The number of arthropod individuals was higher in Rock Creek than in all habitat comparisons. Finally, the mean live weights for female voles in fall was much higher in OU1 than in Rock Creek, although few females were captured in Rock Creek mesic grasslands.

Woodland habitat vegetation endpoints for OU1 showed more tree and shrub density and standing crop than Rock Creek sites, although species richness was less. Review of standard deviations indicates no means significantly different between areas. Small mammal sampling in spring showed no significant differences. Fall sample weights for deer mice and meadow voles showed large differences. Specifically, live weights for adult female deer mice in fall were greater in OU1 than Rock Creek sites and live weights for both sexes of meadow voles in fall were greater in OU1. The difference in weights of female deer mice is explained by the higher ratio of pregnant females than those captured in Rock Creek. The small mammal weight data indicates good health of the small mammal community for OU1 woodland habitats.

Marsh habitat communities showed some differences in both plant and animal endpoints. Vegetation species richness was less in OU1, although standing crop was greater when compared

to Rock Creek sites. These facts are explained by the structural differences between the two marsh areas as discussed in the beginning of this section. OU1, with more cattails and open water, would inherently show less species richness (only two species of cattails on the entire plant site) and more variability in standing crop when compared to the short marsh type (bulrush) community complex in Rock Creek. Spring small mammal sampling showed abundance (number of individuals) and species richness was higher in OU1. Fall sample live weights for both sexes of deer mice and meadow voles were also higher in OU1. These values for small mammals show the dynamic nature of rodent populations (e.g., weights of female voles were higher in Rock Creek sites during spring, then higher in OU1 sites during fall).

All endpoint differences for these habitat comparisons are explainable by site differences, vegetation structure (in the case of marshlands), and dynamics in rodent populations. Therefore, OU1 appears a healthy ecosystem and reflects no stress associated with chemical contaminants.

## E5.0 SUMMARY

The principal objective of the remedial investigation at RFP is to collect data necessary to determine the nature, extent, distribution, and migration pathways of contaminants within the OU1 study area that have the potential to cause adverse ecological impacts. The responsibility of Appendix E, the EE for OU1, is to determine if COCs that are potentially the result of plant releases are producing adverse ecological impacts.

The ecological evaluation in this report is conducted using a three step process. The first step (screening) consists of identifying potential COCs that are known to be present at OU1. The second step, ecological risk assessment (characterizing potential impacts), consists of reviewing research and regulatory findings to determine potential toxicity and behavior of COCs in the soils, surface waters, and sediments. The third step, ecological comparison (actual ecological impacts), consists of evaluating ecological data from RFP to determine if measurable ecological impacts result from COCs at the OU1 study area.

The EE consists of an ecological risk assessment and ecological comparisons. A five-step process for the ecological risk assessment was used, while ecological comparisons consisted of two components. The ecological risk assessment steps were data collection, data evaluation, toxicity assessment, exposure assessment, and risk characterization. The ecological evaluation consisted of species diversity and trophic level comparisons between the OU1 study area and Rock Creek watershed, the reference area.

The period of chemical and biological data collections at RFP for the OU1 EE was January 1991 through March 1992. A total of 139 biological tissue samples were collected to determine if COCs were bioavailable to the ecosystem. Ecological samples of plant and animal taxa, populations, and communities reported 399 taxa. There were 219 species of plants and 180 taxa of animals within the OU1 study area. The most important factor affecting species diversity in communities at RFP is the amount of moisture available to support plant growth, the primary producers in the food web, and food for animals.

Results of the ecological risk assessment section are presented under three headings—toxicity assessment, exposure assessment, and risk characterization.

The toxicity assessment considers chemicals at OU1 that present a significant threat to ecological receptors and evaluates their potential toxicity. General toxicity information on each COC is used to develop TRVs and FRVs for comparison with actual and estimated exposures at OU1. The first stage screening of COCs included heavy metals, cyanide, and radionuclides (because of their high profile at the site) that have been detected above background in soils, surface water, or sediments at OU1. Soils, surface water, and sediments are considered the main pathways for direct exposure of ecological receptors to contaminated media. The second stage screening process was based primarily on exceedance of RFP background concentrations, and secondarily on relative toxicity and bioavailability. The screening process identified chromium, lead, and zinc for soils; chromium, lead, and mercury for surface water; and no COCs were identified for sediments. Radionuclide levels in environmental media was of no concern to ecological receptors because of the very low concentrations.

The exposure assessment is conducted to estimate the magnitude of actual or predicted exposure concentrations and pathways by which ecological receptors are potentially exposed to contamination from the COCs occurring at IHSSs within the OU1 study area. Several very restricted areas have elevated concentrations of heavy metals in the soil. Most metals did not exceed background concentrations by more than twofold and probably do not represent contamination from releases at RFP. The background concentration of COCs at RFP was assumed to be below the toxicity threshold for metals of ecological receptors. Moreover, the exposure assessment failed to provide clear evidence of transport of contaminants away from the localized source areas at IHSSs. However, there was some evidence that sources in upgradient areas impacted water and sediment chemistry in areas adjacent to and downgradient from OU1. The upgradient areas having the highest metal concentration were natural ground-water seeps affected by the highly mineralized bedrock.



The risk characterization summarizes and combines outputs of the toxicity and exposure assessment sections to estimate the relative risk from exposure to COCs at OU1, both in quantitative expressions and qualitative statements. The risk characterization focuses on toxicological risks to ecological receptors from exposure to chromium, lead, zinc, and mercury, and the aggregate risk of simultaneous exposure to these COCs.

Chromium risks were assessed for soils and surface water. The HQ method indicated a moderate risk level associated with the highest chromium concentrations in IHSSs 119.1 and 119.2 for soils and low risk level for all other sites in the OU1 study area. The overall risk to ecological receptors from exposure to chromium in soils in the OU1 area is low. Chromium was above background levels in surface water in only a single sample from sample station SW070. No samples exceeded the (chronic) Colorado Surface Water Quality Standard for total chromium of 170 µg/l. The restricted distribution of chromium at OU1 and lack of downgradient contamination indicate low risk to ecological receptors from exposure, and little or no risk of off-site transport.

Lead risks were assessed for soils and surface water. Lead was detected at concentrations above background in soils at four sites within and around IHSSs 119.1 and 119.2. Concentrations at three of the sites were approximately 10 percent above background. The fourth sample from IHSS 119.1 contained lead at 228 mg/kg, greater than fourfold the RFP background; a second sample from the same site contained only 78 mg/kg. These data suggest that lead contamination in soils within IHSS 119.1 at OU1 is highly localized. The HQ value for exposure to lead by vegetation and soil invertebrates indicated moderate risk. Without the isolated high lead concentration in soil the mean lead value in the OU1 study area is in the range of RFP background concentrations. The HQ method indicated low risk to ecological receptors from lead in soils at OU1 because of the restricted distribution. In surface water lead was above background for Woman Creek upgradient and downgradient of the OU1 study area. The highest lead concentration in the Woman Creek drainage was 13.2 µg/l from a tributary south of the plant site and upgradient of OU1 study area. The South Interceptor Ditch had the overall highest lead

concentration, 38.4 µg/l (dissolved); however, lead levels of this magnitude were detected only once. All other samples from the same station in the South Interceptor Ditch contained lead concentrations only slightly above the RFP background level. The HQ value for the highest lead concentration in surface water from the single sample in the South Interceptor Ditch corresponds to high environmental risk. However, the overall risk to ecological receptors due to lead exposure in OU1 is considered low. This evaluation is based on the generally low level of lead in soils, surface water, and sediments, and the restricted distribution of areas with elevated concentrations. Lead concentration in biological tissue from OU1 indicate minimal uptake of lead.

Zinc risks were assessed for soils at OU1 where the distribution was similar to that of chromium. The average zinc concentration was near background with only 3 of 28 sample sites exceeding twofold the background concentration at RFP. The highest zinc concentrations were detected in IHSS 119.2 and around Building 881. The mean zinc concentration of the three highest values was  $70.73 \pm 15.2$  mg/kg which is near the background level ( $61.98 \pm 12.74$  mg/kg). None of the zinc HQ values for soil indicated risks above the low level. The overall HQ values for vegetation and soil invertebrates also are at the low risk level.

Mercury risks were assessed for surface water at OU1. Mercury concentrations did not exceed background in OU1 soils, surface water, or sediments in Woman Creek downgradient from OU1 IHSSs. Mercury concentration in surface water samples did exceed background in two sites on branches of Woman Creek that drain areas south of the RFP industrial area. However, these areas are outside potential impact from OU1 areas and the concentrations exceeded background by less than 30 percent. Mercury concentrations did exceed background in surface water samples at two stations in the South Interceptor Ditch, however, the source is likely upgradient of OU1 sources and the water from the South Interceptor Ditch, a ground water collection system, is not discharged into surface streams without treatment. The highest concentration of mercury in surface water from the single station in the South Interceptor Ditch was 1.0 µg/l; the HQ value indicates a moderate ecological risk. The risk from mercury in surface water at other stations

is low. The acute and chronic Colorado Water Quality Standards for mercury are 2.4 and 0.1 µg/l, respectively. Therefore, the overall risk to ecological receptors from mercury exposure at OU1 is judged to be low. This assessment is due to the infrequent exceedances of background levels in surface water and the lack of evidence for contamination in other media, including biological tissues collected from the site.

The sum of the mean HQ values (HI) for exposure of vegetation and soil invertebrates was used to assess the overall risk attributed to soils in OU1 IHSS and non-IHSS areas. The HI values for vegetation in IHSS areas, non-IHSS areas, and background areas were 3.4, 2.4, and 2.5, respectively; the values for the same components for soil invertebrates were 2.5, 1.8, and 1.9, respectively. This suggests that vegetation growing in the IHSS areas may be at moderate risk from exposure and that vegetation in non-IHSS areas, and soil invertebrates in all areas are at low risk. For both taxonomic groups, the HI for non-IHSS areas in OU1 was not different from that calculated for the background areas in the Rock Creek drainage.

The risks to ecological receptors from exposure to multiple contaminants in aquatic habitat is also low. Mercury and lead were the only analytes to exceed background by more than twofold at any surface water or sediment station downgradient of OU1 source areas. Their distribution was restricted to the South Interceptor Ditch which was constructed and is maintained as a wastewater collection system. The South Interceptor Ditch provides limited and low quality aquatic habitat because of frequent absence of surface water. The South Interceptor Ditch drains to Pond C-2, the terminal pond in its path. Data from the surface water monitoring program did not indicate concentration of the COCs exceeded RFP background. It should be noted that data on sediments were not available for Pond C-1 or C-2 and, therefore, information on contamination of sediments in the ponds is lacking.

Both the ecological risk characterization and ecological comparisons sections of this report focus on the toxicity threshold for sensitive organisms (i.e., the concentration of chemicals that produce measurable toxic effects on ecological receptors). Since each species has a different sensitivity

threshold, increasing the concentration of toxic chemicals selectively impacts intolerant species. This is reflected by reductions in species diversity (richness), as well as occasional increases in abundance of the remaining tolerant species because of reduced competition for food and habitat. Food webs respond in a similar manner by loss of species performing functions at impacted trophic levels. An ecosystem may become dysfunctional if chemical concentrations are such that species are eliminated.

Ecological comparisons at RFP were conducted at two organizational levels for the OU1 ecosystem—taxonomic structure and trophic function. The taxonomic structure, species diversity, is widely used to compare study areas with similar "nonimpacted" reference areas to determine if pollutional stress is affecting the species composition of communities. As environmental stress from pollutants is gradually increased, the number of species decreases in response to surpassing toxic thresholds for individual species. Since each species occupies a niche, the ecological role or trophic function, then impacts are measurable also by trophic level comparisons. Together the taxonomic and trophic level comparisons provide a yardstick to assess the health of an ecosystem.

The taxonomic group comparisons provided an estimate of the ecosystem's general health based on species diversity within the OU1 study area. The terrestrial ecosystem revealed no difference between the percentage of small mammalian species at OU1 and the Rock Creek reference area. The highest difference was only 3 percent, indicating similar species richness in the two areas. Small mammals are very sensitive indicators of stress caused by COCs entering the food pathway because they are primarily omnivores and herbivores and live in close contact with the soil, the major exposure point in the OU1 study area. The aquatic ecosystem showed slight differences in species richness for plankton and benthic macroinvertebrates. Rock Creek had 11 percent more plankton species and 9 percent less benthic macroinvertebrate taxa than the OU1 study area. These differences were expected since the semiarid climate caused both streams to have recurring loss of habitat during intermittent flow. In particular, Rock Creek has a smaller watershed area upgradient of RFP contributing to the base flow. The comparison of species richness for the OU1 study area and Rock Creek indicates similar levels of ecosystem health. Statistical

evaluations, using chi-square calculations, showed no significant difference between OU1 and Rock Creek for terrestrial or aquatic taxonomic richness.

The species used in the taxonomic level comparison were organized by trophic levels for a food web comparison. Terrestrial arthropods were compared separately since a less detailed identification endpoint was selected for study objectives. As in the taxonomic comparisons, trophic comparisons revealed much similarity between the OU1 and the Rock Creek ecosystems. The maximum percentage difference for any trophic level between these two areas was approximately 4 percent. Comparisons of terrestrial arthropods showed differences of only 6 percent or less between the two areas.

Trophic level comparisons for aquatic primary producers and omnivores showed differences up to 10 percent between OU1 streams and Rock Creek. The comparison of aquatic primary producers does not differ from the plankton comparison. Comparison of species richness for omnivores at the two areas revealed greater species richness at OU1, reflecting good ecosystem health in the OU1 study area.

Chi-square calculations for terrestrial trophic levels revealed no significant difference between areas, again indicating much similarity between areas. Aquatic trophic levels, however, did show a significant difference between areas. This is due to the total number of aquatic taxa being higher in OU1 streams than in Rock Creek, although primary producers (algae) were higher in Rock Creek. Rock Creek tends to be more intermittent than Woman Creek, and therefore is a more harsh environment for aquatic organisms, explaining the diminished number of taxa at the Rock Creek reference area when compared to the OU1 area in Woman Creek watershed.

Results for food web comparisons did not indicate a stressed ecosystem, nor did they reveal ecological problem areas. These results, along with those of tissue sample analysis, should be viewed as further weight-of-evidence indicating an ecologically healthy state at the OU1 study area.

Plant and animal endpoints were compared by habitat type using the mean, standard deviation of a population, and a standard error to compare ecological habitat quality. While reviewing this endpoint data, three general sources of variability became apparent. The first, dealing with the xeric habitat sites, is that only one sample site for OU1 could be established within the ecological study area boundary. Four sites were established in the Rock Creek reference area, this being the normal protocol. The low number of replicates available in OU1 causes variability in the data in that the reference results were derived from four times the number of observations. In addition, the soil in the single OU1 xeric site was extremely compacted, which physically inhibits plant growth and small mammal burrowing. Second, the marsh habitats in OU1 and Rock Creek comprise different wetland plant types. The OU1 marsh habitats are the result of construction activities that created the South Interceptor Ditch and Ponds C-1 and C-2. These areas contain mainly cattails and open water. The vegetative structure differs from marsh habitat found in Rock Creek, which are mostly the result of ground-water seeps and are more suitable habitat for rushes and wet grass species. Although both areas are hydric in nature, these site differences explain the considerable variation of species richness and productivity of both plants and animals. Finally, results for ecological comparisons do not indicate a difference that may be the result of contaminants.

## E6.0 CONCLUSIONS

The results and analysis of EE data contribute the following six conclusions:

- Contaminated areas are very restricted in size, and the concentrations of COCs detected were nonhomogeneous. Some sample stations had high concentrations of COCs in one sample, while other samples indicated only background levels. Therefore, any remedial actions, if needed, would be minor.
- Chromium, lead, mercury, and zinc were identified as COCs that were found in the OU1 study area in concentrations greater than twofold above background RFP values. These COCs potentially were released by site activities, but were ascertained to pose low risk to ecological receptors.
- IHSSs 119.1 and 119.2 areas have the highest levels of COCs for chromium, lead, and zinc.
- There is low bioaccumulation of COCs in ecological receptors at OU1.
- Ecological comparisons at OU1 show no significant difference between species diversity at OU1 and Rock Creek, the reference area. Therefore, there is no indication of adverse ecological impacts as a result of RFP activities.
- The minor ecological differences found between the OU1 study area and Rock Creek were the types of stresses caused by the semiarid climate and not by metal contamination.
- There is low overall risk to ecological receptors at OU1 from COCs.

## E7.0 REFERENCES

- Anderson, S.H. and R.I. Van Hook, Jr. 1973. *Uptake and biological turnover of Cd in chipping sparrows, Spizella passerina*. Environ. Physio. Biochem. 3:243-247.
- Bartlett, R.D. and D.C. Riego. 1972. *Toxicity of hydroxy aluminum in relation to pH and phosphorus*. Soil Science 114:194-200.
- Beyer, W.N. O.H. Pattee, L. Sileo, D.J. Hoffman, and B.M. Mulhern. 1985. *Metal contamination in wildlife living near two zinc smelters*. Environ. Pollut. 38A:63-86.
- Codes of Colorado Regulation (CCR). 1989. *Colorado Water Quality Standards 3.1.0* (5 CCR 1002-8) as amended Sept. 30, 1989.
- Colorado Department of Health (CDH). 1990. *Final Clear Creek Phase II Remedial Investigation*.
- Demayo, A., M.C. Taylor, K.W. Taylor, and P.V. Hodson. 1982. *Toxic effects of lead and lead compounds on human health, aquatic life, wildlife, plants, and livestock*. CRC Crit. Rev. Environ. Control 12:257-305.
- Denenberg, V.H. 1976. *Statistics and Experimental Design for Behavioral and Biological Researchers*. John Wiley & Sons, New York. 344pp.
- Department of Energy (DOE). 1992a. *Final Ground Water Assessment Plan*, Rocky Flats Plant, U.S. Department of Energy, Golden, Colorado, December 1991.
- DOE. 1992b. *Final Historical Release Report for the Rocky Flats Plant*, Golden, Colorado, June 1992.
- DOE. 1992c. *Baseline Biological Characterization of Terrestrial and Aquatic Habitats at Rocky Flats Plant*, Golden, Colorado, September 1992.
- DOE. 1991a. *Final Phase III RFI/RI Work Plan*, Rocky Flats Plant, 881 Hillside Area (Operable Unit No. 1), U.S. Department of Energy, Golden, Colorado, March 1991.
- DOE. 1991b. *Final Phase III RFI/RI Environmental Evaluation Work Plan: Field Sampling Plan, Rocky Flats Plants 881 Hillside Area (Operable Unit No. 1)*, U.S. Department of Energy, Golden, Colorado, June 1991.
- DOE. 1991c. *EMAD Operating Procedures Manual Volume V: Ecology*. Manual No. 5-21200-OSP-EE. August 1991.



including grasshoppers, beetles, earwigs, springtails, bugs, flies, and ants. Zinc concentration in these primary consumers was similar to that found in cover vegetation (220 µg/g; control levels approximately 55 µg/g). Zinc concentration increased in such secondary consumers as spiders, beetles, and centipedes (280 µg/g; controls at 50 µg/g).

In five species of phytophagous insects (three Hymenoptera and two Lepidoptera), Zn concentrations were much greater than any other metal studied, and 100 times higher than Cd concentrations (Lindqvist 1992). In all cases, body levels of Zn were higher in larvae than in the food plants they consumed and higher in adults than in larvae.

Earthworms of three genera (*Lumbricus*, *Alabophora*, and *Octoclasium*) from six undisturbed soil types showed concentration ratios of 3 to 13 (Van Hook 1974). Typically, concentration ratios are higher in worms from less polluted soils, reinforcing the idea that many organisms actively control accumulation of this metal. In *Lumbricus terrestris*, tissue concentrations correlated with distance from an old mine site (highest Zn concentration =  $210.0 \pm 37.2$  µg/g; control =  $90.0 \pm 7.02$  µg/g). *Lumbricus rubellus* taken directly from the soil at an abandoned mine showed much higher Zn concentrations:  $2,511.4 \pm 305.6$  µg/g ( $426.1 \pm 34.2$  µg/g at control site). When localization studies were done, the greatest concentration of Zn was observed in the posterior alimentary canal (Morgan and Morgan 1990).

Worms of three genera (*Lumbricus*, *Dendrobaena*, and *Octoclasium*) from a smelter site showed a strong correlation between body weight and Zn concentration; the highest correlation was seen closest to the smelter (C.H. Jones, unpub., reported in Martin and Coughtrey 1982). Population densities of nine worm species from two genera (*Lumbricus* and *Allolobophora*) were lower at a smelter site than at a control site (64 worms/m<sup>2</sup> versus 161.8 worms/m<sup>2</sup>). Zinc concentrations in worms ranged from 634 - 1,398 at the smelter and 264 - 914 at the control site; concentration factors ranged from 1.03 - 2.26 at the smelter and 2.97 - 10.25 at the control site (Wright and Stringer 1980).

- EPA. 1989b. *Ecological Assessments at Hazardous Waste Sites: A Field and Laboratory Reference Document*, March 1989. EPA/600/3-89/013.
- EPA. 1989c. *Ecological Risk Assessment Methods: A Review and Evaluation of Past Practices in the Superfund and RCRA Programs*, June 1989. EPA/230/3-89/044.
- EPA. 1989d. *Supplemental Risk Assessment Guidance for the Superfund Program*. Draft Final, June 1989.
- EPA. 1989e. *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual Part A*. Interim Final, December 1989. EPA/540/1-89/002.
- EPA. 1988. *Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA*. EPA/540/G-89/004.
- EPA. 1987. *Data Quality Objectives for Remedial Response Activities Development Process*, March 1987. EPA/540/G-87/003.
- EPA. 1986. *Hazard Evaluation Division Standard Evaluation Procedure Ecological Risk Assessment*. EPA/540/9-86/167.
- EPA. 1985a. *Ambient Water Quality Criteria for Arsenic*. Office of Water Regulations and Standards. 440/5-84/003.
- EPA. 1985b. *Ambient Water Quality Criteria for Lead*. Office of Water Regulations and Standards. 440/5-84/027.
- EPA. 1985c. *Ambient Water Quality Criteria for Chromium*. Office of Water Regulations and Standards. 440/5-84/029.
- EPA. 1985d. *Ambient Water Quality Criteria for Copper*. Office of Water Regulations and Standards. 440/5-84/031.
- EPA. 1985e. *Ambient Water Quality Criteria for Cadmium*. Office of Water Regulations and Standards. 440/5-84/032.
- EPA. 1985f. *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*.
- EPA. 1980. *Ambient Water Quality Criteria for Silver*. Office of Water Regulations and Standards. 440/5-80/071.

- Fordham, C.L. and D.P. Reagan. 1991. *Pathways analysis method for estimating water and sediment criteria at hazardous waste sites*. Environ. Toxicol. and Chem., 10:949-960.
- Fowler, B.A. et al. 1977. *Ultrastructural and biochemical effects of prolonged oral arsenic exposure on liver mitochondria of rats*. Environ. Health Perspect. 19:197.
- Goyer, R.A. 1986. *Toxic effects of metals*. In *Toxicology: The Basic Science of Poisons* (C.D. Klaassen, M.O. Amdur, and J. Doull, eds.). MacMillan Publishing Company, New York.
- Horst, W.J. 1985. *Effect of aluminum on root growth, cell division rate and mineral element contents in roots of Vigna unguiculata genotypes*. Z. Pflanzenernahr. Bodenk 148:335-348.
- Interagency Agreement (IAG). 1991. *Rocky Flats Interagency Agreement between the State of Colorado, the Environmental Protection Agency, and the Department of Energy*, January 1991.
- International Commission on Radiological Protection (ICRP). 1960. *Report on Committee II on permissible dose for internal radiation (1959)*. Health Phys. 3:1.
- Jernelov, A., K. Beijer, and L. Soderlund. 1978. *General aspects of toxicology*. In *Principles of Ecotoxicology* (G.C. Butler, ed.). SCOPE 12. John Wiley & Sons, New York.
- Johnson, M.S., R.D. Roberts, M. Hutton, and M.J. Inskip. 1978. *Distribution of lead, zinc, and cadmium in small mammals from polluted environments*. Oikos 30:153-159.
- Lewis, S.C., J.R. Lynch, and A.I. Nikiforov. 1990. *A new approach to deriving community exposure guidelines from "No-Observed-Adverse-Effects-Levels."* Reg. Toxicol. and Pharmacol. 11:314-330.
- Martin, M.H. and P.J. Coughtrey. 1982. *Biological Monitoring of Heavy Metal Pollution*. Applied Science Publishers, London. 475pp.
- Moriarty, F. 1983. *Ecotoxicology: The Study of Pollutants in Ecosystems*. Academic Press, New York. 289pp.
- National Academy of Science (NAS). 1973. *Manganese*. Washington, D.C. 108pp.
- Peltier, W. and C.I. Weber, eds. 1985. *Methods for Measuring the Acute Toxicity of Effluents to Freshwater and Marine Organisms*. Third Edition. Environmental Monitoring and Support Laboratory, U.S. EPA, Cincinnati. EPA 600/4-85-013. 230pp.
- Rand, G.M. and S.R. Petrocelli. 1985. *Fundamentals of Aquatic Toxicology: Methods and Applications*. Hemisphere Publishing Corp., Washington, D.C. 666pp.

Woodlice *Oniscus asellus* are important detritus feeders in many ecosystems. Like snails and slugs, they typically bioconcentrate Zn above levels in soil or on vegetation near contaminated sites (BCFs of 1.37 - 1.63), although in litter the ratios were less than one (0.05 - 0.30) (Martin and Coughtrey 1976). In one instance, the concentration of Zn in decaying litter near a Zn smelter was very similar to that found in the isopods. The mean Zn concentration from the isopods *Porcellio scaber* and *O. asellus* from a variety of sites was  $1.47 \pm 0.09$   $\mu\text{g/g}$  (range 0.17 - 7.41  $\mu\text{g/g}$ ). The mean concentration ratio for these animals was  $0.75 \pm 0.06$  (range 0.08-3.03) (Martin and Coughtrey 1982). A strong correlation was seen between Zn concentration and body weight,  $r=0.76$ .

Like other organisms, isopods probably actively regulate endogenous Zn levels (Coughtrey et al. 1980). Unlike many arthropods, isopods do not store Zn in their exoskeleton. The hepatopancreas is the primary storage organ for Zn in isopods, where up to 76 percent of the Zn may be held. In isopods taken from many sites, including old mines, the maximum Zn concentration was found to be  $178 \pm 14$  ppm (54.3 at control sites); no toxic effects were seen in any of these animals (Hopkin and Martin 1982).

Mosquito (*Aedes aegypti*) larvae treated in experimental flasks containing 0 - 50 ppm Zn experienced no mortality after 48 hours (Abbasi et al. 1988). Pupae placed in flasks containing Zn solutions exhibited 20 percent mortality after 48 hours at 0.5 ppm and 30 percent mortality after 48 hours at 5 ppm (Abbasi and Soni 1983). Despite the relatively high survivorship in both treatment groups, behavioral and/or physiological abnormalities that prevented them from swimming or flying as adults would have resulted in much higher mortality under natural conditions.

### 14.3 TERRESTRIAL VERTEBRATES

Beyer et al. (1985) found that very little of the Zn in soil was incorporated in flora and fauna; contamination came predominantly from aerial deposition. They also found higher concentrations of Zn in shrews and lower concentrations in mice, in contrast to Roberts and Johnson (1978),

## Appendix E • Tables

Environmental  
Evaluation

Phase III  
RFI/RI Report

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Table E 1.4-1 Sampling Design Matrix for OU1 Ecological Studies

Taxa Sampled	Measurement Endpoints	Season of Sampling	Diurnal Constraints	Number of Sites To Be Sampled	Number of Sites Sampled	Configuration of Area Sampled At Site
All Terrestrial Plants	Areal extent	-	-	ESA	ESA	ESA
	Mapped location	Su, Fa	-	ESA	ESA	ESA
	Species presence	Sp, Su, Fa	-	ESA and all belt transects	ESA & 10 belt transects	ESA & 50 x 2 transects
	Species richness	LSp, ESu	-	ESA and all belt transects	ESA & 10 belt transects	ESA & 50 x 2 transects (transects nested in small mammal plots)
Herbaceous Plants	Cover by species (including bare ground, rock, lichen, litter)	LSp-ESu, Fa	-	10 St and 10 Rf transects in each habitat, LSp-ESu (except St only in Rehabilitation and Disturbance)	10 St and 10 Rf transects in each habitat, LSp-ESu (except St only in Rehabilitation and Disturbance)	100 points in every 50 x 2 m transect (10 points every 5 m in LSp-ESu; 1 point every 50 cm in Fa) (transects nested in small mammal plots)
	Standing Crop	Fa	-	80 % cert., max. of 30	30 Marshland St transects, Fa 30 Marshland Rf transects, Fa 30 Disturbance St transects, Fa 30 Woodland St transects, Fa 30 Woodland Rf transects, Fa 30 Mesic Mixed Grass St transects, Fa 30 Mesic Mixed Grass Rf transects, Fa 20 Rehabilitation St transects, Fa 30 Xeric Mixed Grass St transects, Fa 20 Xeric Mixed Grass Rf transects, Fa	10-0.25 m.sq. plots per cover transect
Low Shrubs	Standing Crop	Fa	-	At all herbaceous plant standing crop locations	At all herbaceous plant standing crop locations	10-0.25 m.sq. plots per cover transect EERFP (10/92)

Table E 1.4-1 Sampling Design Matrix for OU1 Ecological Studies

Taxa Sampled	Measurement Endpoints	Season of Sampling	Diurnal Constraints	Number of Sites To Be Sampled	Number of Sites Sampled	Configuration of Area Sampled At Site
Shrubs and Trees (including subshrubs, cacti and yucca)	Density Canopy cover	LSp, ESu, Fa	-	As for herbaceous cover	As for herbaceous cover	As for herbaceous cover
All Wildlife	Relative abundance	Sp, Su, Fa, Wi	-	3 transects (run 1 time/season, more often in spring)	3 transects (each run 2 times in spring, 3 times in summer, 2 times in fall; continuing)	40 minute traverse is minimum line length
Arthropods	Taxon presence Taxon richness Taxon relative abundance	Sp, Fa	Heat of day	3 transects per habitat as available; each sampled 3 times	3 Marshland St transects 3 Marshland Rf transects 2 Disturbed St transects 3 Mesic Mixed Grass St transects 3 Mixed Grass Rf transects 3 Reclamation St transects 3 Woodland St transects 3 Woodland Rf transects 1 Xeric Mixed Grass St transect 3 Xeric Mixed Grass Rf transects Each of the above sampled 3 times Night sampling 1 time, briefly at 18 locations	50 x 2 m transect adjacent to plant cover transects in small mammal plots
Birds	Species presence Species richness Species relative abundance Breeding species relative density/ha	Sp, Su, Fa, Wi Sp, Su, Fa, Wi Sp, Su, Fa, Wi LSp, Su, Fa (ongoing)	- - - Before 10:00 am	See relative abundance transects See relative abundance transects See relative abundance transects 1000 m St and Rf transect in each habitat (surveyed 3 times)	See relative abundance transects See relative abundance transects See relative abundance transects 1000 m Reclamation St transect across 4 sites 200 m Disturbed St transect across 1 site 100 m Xeric Grass St transect across 1 site 1000 m Xeric Grass Rf transect across 2 sites 1000 m Mixed Grass St transect across 3 sites 1000 m Mixed Grass Rf transect across 1 site 1000 m Woodland St transect across 1 site 1000 m Woodland Rf transect across 1 site	See relative abundance transects See relative abundance transects See relative abundance transects 1000 m subdivided into 100 m segments, sometimes contiguous, sometimes not

EERFP (10/92)

Table E 1.4-1 Sampling Design Matrix for OU1 Ecological Studies

Taxa Sampled	Measurement Endpoints	Season of Sampling	Diurnal Constraints	Number of Sites To Be Sampled	Number of Sites Sampled	Configuration of Area Sampled At Site
Birds (Cont.)					1000 m Marshland St transect across 1 site 1000 m Marshland Rf transect across 1 site each of above surveyed 3 times	25 trap locations per plot, 5 m apart, but in 1 to 5 lines
Small Mammals	Species presence Species richness Species relative abundance	Sp, Fa	Set traps late afternoon Check traps early morning	4 plots per habitat (sampled for 4 nights)	4 St and 4 Rf plots in Aquatic, Grassland, Woodland habitats; 4 Reclamation St plots; 2 Disturbed St plots; 1 Xeric Grass St plot and 4 Xeric Grass Rf plots; see Table 2 for specific configuration	depending on habitat configuration See relative abundance transects See relative abundance transects See relative abundance transects
Large Mammals	Species presence Species richness Species relative abundance Ungulate species pellets per ha	Sp, Su, Fa, Wi Sp, Su, Fa, Wi Sp, Su, Fa, Wi Fa, Sp 1992	- - - -	See relative abundance transects See relative abundance transects See relative abundance transects 25 circular plots in each of 35 small mammal plots	See relative abundance transects See relative abundance transects See relative abundance transects 25 circular plots in each of 35 small mammal plots	1.79 m circle at each small mammal trap location Entire pond Entire pool 50 m up and downstream from selected riffle site Entire seep
Plankton	Taxon presence Taxon richness Taxon relative abundance Taxon relative density/ml Ash-free dry weight & volume	Fa	-	2 discharge St ponds, 2 other Rf ponds	1 liter grab sample at each of 5 locations in each pond	
Periphyton	Taxon presence Taxon richness Taxon relative abundance Taxon relative density/mm.aq. Chlorophyll a/phaeophytin a	Fa, Sp 1992	-	2 discharge St ponds, 2 other Rf ponds; 3 St riffles, 2 Rf riffles; 2 St pools, 2 Rf pools; 1 St seep, 1 Rf seeps	2 floating samplers in each pond; riffle and seep site - qualitative	See plankton
Benthic Macroinvertebrates	Taxon presence Taxon richness Taxon relative abundance	Sp, Fa	-	See periphyton	5 samples of 2000 ml bottom in each pond, pool and seep site; 5 Surber samples at each riffle site	See relative abundance transects See relative abundance transects See relative abundance transects
Amphibians	Species presence Species richness Species relative abundance Species relative abundance	Sp, Su, Fa Sp, Su, Fa Sp, Su, Fa Sp	- - - After dark	See relative abundance transects See relative abundance transects See relative abundance transects 2 drainages (Woman and Rock Creeks)	See relative abundance transects See relative abundance transects See relative abundance transects None due to late start and lack of timely nighttime access permission	See plankton EERFP (10/92)



Table E 1.4-1 Sampling Design Matrix for OU1 Ecological Studies

Taxa Sampled	Measurement Endpoints	Season of Sampling	Diurnal Constraints	Number of Sites To Be Sampled	Number of Sites Sampled	Configuration of Area Sampled At Site
Fishes	Species presence Species richness Species relative abundance	Sp, Fa	-	See periphyton	Minimum of 2 gill-net-nights in each pond; minimum of 6 minnow-trap-nights in each pool, rifle and seep site	

ESA=Entire Study Area

E=Early, L=Late

St=Study Sites, Rf=Reference Sites

Sp = Spring (Mar.-May)

Su = Summer (Jun.-Aug.)

Fa = Fall (Sep.-Nov.)

Wi = Winter (Dec.-Feb.)

LSp = Late Spring

LSu = Late Summer

Table E.3.5-1. Chemicals Selected as Contaminants of Concern for the OU1 EE

<u>Metals</u>	<u>Radionuclides</u>	<u>Inorganics</u>
aluminum (Al)	americium-241 (Am)*	cyanide (CN)
arsenic (As)	plutonium-239 (Pu)*	
beryllium (Be)	radium-226 (Ra)	
cadmium (Cd)	strontium-90 (Sr)*	
chromium (Cr)	uranium (total) (U)*	
copper (Cu)	gross alpha <sup>1</sup>	
iron (Fe)	gross beta <sup>1</sup>	
lead (Pb)*		
manganese (Mn)	<hr/> total = 5	
mercury (Hg)		
nickel (Ni)*		
silicon (Si)*		
silver (Ag)		
zinc (Zn)		
<hr/> total = 14		

\* Not identified as a COC in OU1 FSP

<sup>1</sup> Total particle counts

Table E 4.1-1 Toxicity Reference Values (TRVs) for OU1 Environmental Evaluation

## Soil TRVs for Terrestrial Vegetation

Analyte	Species	Endpoint	TRV	units	Reference
Aluminum	NONE				
Arsenic	grasses	Reduced growth	15	mg/kg	National Academy of Sciences. 1977. Arsenic. NAS: Washington D.C.
Beryllium	NONE		2	mg/kg	Wilber, C.G. 1980. Beryllium-A Potential Environmental Contaminant. C.C.Thomas, Springfield, MO.
Cadmium	oats	NOEC for plant weight	10	mg/kg	Adema, D.M. & L. Henzen. 1989. Ecotoxicology and Environmental Safety. 18:219.
Chromium III	oats	NOEC for plant weight	3.5	mg/kg	Adema, D.M. & L. Henzen. 1989. Ecotoxicology and Environmental Safety. 18:219.
Chromium VI	NONE				
Copper	tomato	Reduced growth	16.5	mg/kg	Rhoads, F.M. et al. 1989. J. Environmental Quality.
Cyanide	NONE				
Iron	NONE				
Lead	Cassia sp.	Reduced pollen germination	50	mg/kg	Krishnayya et al. 1986.
Manganese	wheat	NOEC for germination	2	mg/kg	Taylor, G.J. et al. 1991. Canadian J. Botany. 69:359.
Mercury	NONE				
Silver	NONE				
Zinc	Bouteloua sp.	Reduced growth	40	mg/kg	White, E.M. 1991. J. Range Management. 44:48.

## Soil TRVs for Terrestrial Arthropods

Analyte	Species	Endpoint	TRV	units	Reference
Aluminum	NONE				
Arsenic	NONE				
Beryllium	NONE				
Cadmium	aggregate species	Absence in soils	11	mg/kg	Eisler, R. 1985. Contaminant Hazard Reviews Report No. 2. USFWS.
Chromium III	Octochaetis sp.	Lethality	2	mg/kg	Abassi, S.A. & R. Sorli. 1983. International J. Environmental Studies
Chromium VI	NONE				
Copper	Eisenia sp.	Reduced growth	55	mg/kg	Hartenstein, R. et al. 1980. J. Environmental Quality. 9:23
Cyanide	NONE				
Iron	NONE				
Lead	NONE				
Manganese	NONE				
Mercury	Octochaetis sp.	Lethality	0.2	mg/kg	Abassi, S.A. & R. Sorli. 1983. International J. Environmental Studies
Silver	NONE				
Zinc	aggregate species	Absence in soils	11	mg/kg	Eisler, R. 1985. Contaminant Hazard Reviews Report No. 2. USFWS.

NOTE: Values used are Colorado State Standards for Agricultural Use of Surface Waters and Protection of Aquatic Life.

Table E 4.1-1 Toxicity Reference Values (TRVs) for OU1 Environmental Evaluation

## Surface Water TRVs for Wildlife and Aquatic Life

Analyte	Wildlife Drinking Water <sup>1</sup>	units	Protection of Aquatic Life <sup>2</sup>	units	Reference
Aluminum	NONE	mg/L	150.00	ug/L	Environment Reporter 726: 1001-1020: 6/1/90)
Arsenic	100.00	mg/L	50.00	ug/L	Ibid
Beryllium	100.00	mg/L	5.30	ug/L	Ibid
Cadmium	10.00	mg/L	1.00	ug/L	Ibid
Chromium III	100.00	mg/L	172.00	ug/L	Ibid
Chromium VI	NONE	mg/L	11.00	ug/L	Ibid
Copper	200.00	mg/L	10.00	ug/L	Ibid
Cyanide	200.00	mg/L	5.00	ug/L	Ibid
Iron	NONE	mg/L	1,000.00	ug/L	Ibid
Lead	100.00	mg/L	3.00	ug/L	Ibid
Manganese	200.00	mg/L	1,000.00	ug/L	Ibid
Mercury	NONE	mg/L	0.10	ug/L	Ibid
Silver	NONE	mg/L	0.20	ug/L	Ibid
Zinc	2,000.00	mg/L	45.00	ug/L	Ibid

<sup>1</sup> Values refer to total recoverable metal<sup>2</sup> Values refer to dissolved metal concentration

Table E 4.1-2. Final Reference Values (FRVs) for OU1 EE Contaminants of Concern : Surface Soils

Analyte	RFP Surficial Background (1992)	Vegetation		Terrestrial Invertebrates		Units
		Reg. Std.	TRV	Reg. Std.	TRV	
Aluminum	16,081	NA	NA	NA	16,081	mg/kg
Arsenic	6.78	NA	1.5	NA	6.8	mg/kg
Beryllium	0.90	NA	2	NA	0.9	mg/kg
Cadmium	0.77	NA	10	NA	11	mg/kg
Chromium	18.3	NA	3.5	NA	18.3	mg/kg
Copper	18.46	NA	17.5	NA	50	mg/kg
Iron	18,961	NA	NA	NA	18961	mg/kg
Lead	40.07	NA	50	NA	40.07	mg/kg
Manganese	435.41	NA	0.5	NA	435.41	mg/kg
Mercury	0.07	NA	NA	NA	0.13	mg/kg
Silver	1.14	NA	NA	NA	1.14	mg/kg
Zinc	71.21	NA	40	NA	1,000	mg/kg
Cyanide	not done	NA	NA	NA	DL	mg/kg
Americium-241	0.023	NA	2.60E+05	NA	3.70E+04	pCi/g
Plutonium	0.048	NA	3.60E+05	NA	3.70E+04	pCi/g
Radium-226	1.13	NA	1660	NA	ND	pCi/g
Strontium-90	not done	NA	2.90E+05	NA	3.70E+04	pCi/g
Uranium (total)	not done	NA	2.90E+05	NA	3.70E+04	pCi/g

Table E.4.1-3. Final Reference Values (FRVs) for Contaminants of Concern: Surface Water (µg/L, except where noted)

COC	Aquatic Life				Wildlife			
	Reg <sup>a,1,2</sup> Std	TRV <sup>2</sup>	RFP <sup>2</sup> Bkgnd	FRV	Reg <sup>a,3</sup> Std	TRV <sup>3</sup>	RFP <sup>3</sup> Bkgnd	FRV
Aluminum	950/150	748	295	485	NA	NA	60,420	60,420
Arsenic	360/50	190	3	50	100	NA	1,030	1,030
Beryllium	130/5.3	5.3	7	5.3	100	NA	10.7	100
Cadmium	7.6/1.0	1	3	5 (DL)	10	NA	64.4	64.4
Chromium III	1450/172	172	Total 10	172	100 <sup>4</sup>	NA	275 <sup>4</sup>	275
Chromium VI	16/11	11	DL	11	---	NA	---	---
Copper	14/10	10	18	28	200	NA	607	607
Cyanide	5	2.5	5 (DL)	5 (DL)	200	NA	5 (DL)	200
Iron	1,000 <sup>3</sup>	53	1,113	16,700	NA	NA	87,150	87,150
Lead	67/3	2.4	4	131	100	NA	516	516
Manganese	1,000 <sup>3</sup>	1,000	292	1,000	200	NA	1,970	1,970
Mercury	2.4/0.1	1.3	0.3	1.3	NA	NA	1.4	1.4
Silver	1.38/0.2	0.12	7	70.2	NA	NA	148	148
Zinc	181/45	47	141	102	2,000	NA	380	2,000
Radium 226 <sup>3,5</sup>	5	0.81	2.3	29.25	5	NA	29.25	29.25
Cesium - 137 <sup>3,5</sup>	80	0.015	.73	80	80	NA	3.93	80
Plutonium 239 <sup>3,5</sup> , 240	15	0.33	.02	15	15	NA	1.46	15
Strontium - 90 <sup>3,5</sup>	8	1.0	1.41	8	8	NA	1.61	8
Uranium <sup>3,5</sup>	NA	21	2.0	21	NA	NA	1.11	NA
Americium - 241	NA	NA	.03	NA	NA	NA	.032	NA

a. CDHWQCC, Colorado Water Quality Standards 3.1.0 (5 CCR 1002-8), amended 9/30/89 (Environment Reporter 726: 1001-1020; 8/1/90)

1. Acute/Chronic
  2. Value for dissolved metals
  3. Value for total recoverable metals
  4. Total chromium
  5. Units = µg/L
- DL = Detection Limit  
NA = Not available

Table E 4.2-1. Species Used for Tissue Sampling

<u>CODE</u>	<u>Latin Name</u>	<u>Common Name</u>
<b>PLANTS</b>		
ARLU	<i>Artemisia ludoviciana</i>	white sage
BOGR	<i>Bouteloua gracilis</i>	blue grama
BRIN	<i>Bromus inermis</i>	smooth brome
MEOF	<i>Melilotus officinalis</i>	yellow sweet clover
POCO	<i>Poa compressa</i>	Canada bluegrass
<b>ANIMALS</b>		
ACRI	Acrididae	grasshoppers
CACO	<i>Catostomus commersoni</i>	white sucker
LECY	<i>Lepomis cyanellus</i>	green sunfish
MIPE	<i>Microtus pennsylvanicus</i>	meadow vole
MISA	<i>Micropterus salmoides</i>	largemouth bass
NOCR	<i>Notemigonus crysoleucas</i>	golden shiner
PEMA	<i>Peromyscus maniculatus</i>	deer mouse
PIPR	<i>Pimephales promelas</i>	fathead minnow
SEAT	<i>Semotilus atromaculatus</i>	creek chub

Table E 4.2-2. Background Concentrations of OU1 Target Metals in Biological Tissues at RFP

Sample Type	Life Form	Study Area Species (1)	Reference Area Species	Community Type	Analyte Concentration (mg/kg dry weight)						
					Cd	Cr (total)	Cu	Pb	Hg	Ag	Zn
Small Mammals (SM)	Mice	PEMA	PEMA	Xeric	3.3	DL	22.5	1.7	DL	DL	117
				Mesic	3.9	4.9	42.8	2.7	DL	DL	146
				Hydric	1.6	DL	17	2.6	DL	DL	106
									DL	DL	
	Voles	MIPE	MIPE	Hydric	3.1	9.7	26.2	1.7	DL	DL	157
									DL	DL	
Terrestrial Arthropods (TA)	Grasshoppers	ACRI	ACRI	Mesic	3.6	DL	73.9	DL	DL	DL	138
				Hydric	3.8	DL	66.3	3.2	DL	DL	144
Vegetation (VE)	---No Reference or Study Samples			Xeric							
	Forbs	ARLU+MEOF	ARLU	Mesic	1.2	1.3	10.7	1.4	DL	DL	44.8
				Hydric	1.1	5.4	11.1	0.9	DL	DL	60.5
									DL		
	Grasses	BOGR+BRIN+POCO	BOGR	Mesic	DL	1.3	4.7	1.4	DL	DL	19.8
			BRIN/POCO	Hydric	DL	1.5	3.1	0.6	DL	DL	19.4
Fish (FI)	Fish	PIPR+NOCR	PIPR	Pond	DL	7.4	40.9	4.3	DL	DL	648
		SEAT+CACO	PIPR	Pond	DL	7.4	40.9	4.3	DL	DL	648
		LECY+MISA	MISA	Pond	2.5	3.8	10.3	1.9	0.5	DL	113

DL=Detection Limit

(1) Key			
PEMA	Peromyscus maniculatus	POCO	Poa compressa
MIPE	Microtus pennsylvanicus	PIPR	Pimephales promelas
ACRI	aggregate Orthoptera	NOCR	Nidropis cornutus
ARLU	Artemisia ludoviciana	SEAT	Semotilus atromaculatus
MEOF	Melilotus officinalis	CACO	Calostomus commersoni
BOGR	Bouteloua gracilis	LECY	Lepomis cyanellus
BRIN	Bromus inermis	MISA	Micropterus salmoides



Site	Species	Analyte	OU1 Sample	RFP Background	Units
MA01A	POCO1	Copper	3.3	1.7	mg/kg
MA01A	POCO1	Lead	0.7	0.6	mg/kg
MA01A	POCO1	Zinc	13.6	12.6	mg/kg
MA02A	MEOF1	Chromium	37.8	0.6	mg/kg
MA02A	MEOF1	Lead	0.7	0.5	mg/kg
MA02A	MEOF1	Total Uranium	1.3	0	pCi/g
MA04A	BRIN1	Chromium	1.9	1.1	mg/kg
MA04A	BRIN1	Lead	0.5	0.4	mg/kg
MD01A	BRIN1	Chromium	1.5	1.3	mg/kg
MD01A	BRIN1	Copper	4.8	4.7	mg/kg
MG02A	ARLU1	Copper	11.4	10.7	mg/kg
MG02A	ARLU1	Zinc	50.9	44.8	mg/kg
MG02A	ARLU1	Zinc	45.6	44.8	mg/kg
MG02A	BOGR1	Chromium	1.7	1.3	mg/kg
MG02A	BOGR1	Chromium	1.4	1.3	mg/kg
MG02A	BOGR1	Copper	14	4.7	mg/kg
MG03A	ARLU1	Cadmium	1.9	1.2	mg/kg
MG03A	ARLU1	Cadmium	1.8	1.2	mg/kg
MG03A	ARLU1	Chromium	2.1	1.3	mg/kg
MG03A	ARLU1	Copper	14.6	10.7	mg/kg
MG03A	ARLU1	Copper	15.2	10.7	mg/kg
MG03A	ARLU1	Zinc	56.4	44.8	mg/kg
MG03A	ARLU1	Zinc	48.1	44.8	mg/kg
MG03A	BOGR1	Chromium	2.1	1.3	mg/kg
MG03A	BOGR1	Chromium	2.2	1.3	mg/kg
MG04A	ARLU1	Chromium	3.4	1.3	mg/kg
MG04A	ARLU1	Copper	16.3	10.7	mg/kg
MG04A	ARLU1	Lead	1.9	1.4	mg/kg
MG04A	ARLU1	Zinc	46.9	44.8	mg/kg
MG04A	ARLU1	Zinc	59.8	44.8	mg/kg
MG04A	BOGR1	Chromium	1.6	1.3	mg/kg
MG04A	BOGR1	Copper	5.4	4.7	mg/kg
MG04A	BOGR1	Zinc	20.7	19.8	mg/kg
MG04A	BOGR1	Radium 226	11	0	pCi/g
MR01A	BRIN1	Chromium	1.5	1.3	mg/kg
MR02A	ARLU1	Chromium	2.7	1.3	mg/kg
MR02A	ARLU1	Copper	12.6	10.7	mg/kg
MR04A	ARLU1	Chromium	1.6	1.3	mg/kg
MR04A	ARLU1	Chromium	1.9	1.3	mg/kg
MR04A	ARLU1	Radium 226	11	0	pCi/g

Table E 4.2-3. Vegetation Tissue Samples Exceeding Background at OU1 Sites Page 2 of 2

Site	Species	Analyte	OU1 Sample	RFP Background	Units
MR04A	BRIN1	Chromium	1.5	1.3	mg/kg
MR04A	BRIN1	Chromium	2	1.3	mg/kg
MR04A	BRIN1	Zinc	24	19.8	mg/kg
MR04A	BRIN1	Zinc	22.5	19.8	mg/kg
MW01A	BRIN1	Cadmium	0.3	0	mg/kg
MW01A	BRIN1	Chromium	34.1	1.1	mg/kg
MW01A	BRIN1	Lead	0.8	0.4	mg/kg
MW01A	BRIN1	Zinc	28.1	19.4	mg/kg
MW03A	ARLU1	Cadmium	1.9	1.1	mg/kg
MW03A	ARLU1	Cadmium	1.6	1.1	mg/kg
MW03A	ARLU1	Copper	11.5	10.2	mg/kg
MW03A	ARLU1	Copper	11.2	10.2	mg/kg
MW03A	POCO1	Cadmium	0.2	0	mg/kg
MW03A	POCO1	Zinc	13	12.6	mg/kg
MW04A	ARLU1	Chromium	23.3	5.4	mg/kg
MW04A	POCO1	Zinc	14.2	12.6	mg/kg

Key	
BRIN	<i>Bromus inermis</i>
ARLU	<i>Artemisia ludoviciana</i>
POCO	<i>Poa compressa</i>
MEOF	<i>Melilotus officinalis</i>
BOGR	<i>Bouteloua gracilis</i>

Table E 4.2-4. Small Mammal Tissue Samples Exceeding Background at OU1 Sites

Site	Species	Analyte	OU1 Sample	RFP Background	Units
MA01A	MIPE1	Lead	2.9	1.7	mg/kg
MA01A/3A	MIPE1	Lead	3.8	1.7	mg/kg
MA02A	PEMA1	Copper	18.7	17	mg/kg
MA02A	PEMA1	Copper	23.8	17	mg/kg
MA03A	MIPE1	Lead	2	1.7	mg/kg
MA04A	PEMA1	Cadmium	1.9	1.6	mg/kg
MA04A	PEMA1	Copper	22	17	mg/kg
MA04A	PEMA1	Zinc	133	106	mg/kg
MD01A	PEMA1	Copper	158	42.8	mg/kg
MD01A	PEMA1	Plutonium 239/240	0.026	0	pCi/g
MD02A	MIPE1	Copper	36.2	26.2	mg/kg
MD02A	MIPE1	Lead	2.3	1.7	mg/kg
MG01A	MIPE1	Cadmium	6.4	3.1	mg/kg
MG01A	MIPE1	Copper	34.4	26.2	mg/kg
MG01A	MIPE1	Lead	6.9	1.7	mg/kg
MG03A	PEMA1	Cadmium	10.6	3.9	mg/kg
MG03A	PEMA1	Copper	43.1	42.8	mg/kg
MG03A	PEMA1	Lead	8.5	2.7	mg/kg
MG03A	PEMA1	Zinc	151	146	mg/kg
MG04A	MIPE1	Lead	1.9	1.7	mg/kg
MG04A	MIPE1	Americium 241	0.086	0	pCi/g
MG04A	MIPE1	Plutonium 239/240	0.47	0	pCi/g
MR02A	MIPE1	Cadmium	4.1	3.1	mg/kg
MR02A	MIPE1	Lead	3.1	1.7	mg/kg
MR02A	MIPE1	Total Uranium	0.26	0	pCi/g
MW01A	PEMA1	Zinc	210	106	mg/kg
MW02A	MIPE1	Lead	1.8	1.7	mg/kg
MW03A	PEMA1	Chromium	3.4	0	mg/kg
MW03A	PEMA1	Copper	17.3	17	mg/kg
MW04A	PEMA1	Copper	28.2	17	mg/kg
MW04A	PEMA1	Lead	2.9	2.6	mg/kg
MW04A	PEMA1	Zinc	153	106	mg/kg

**Key**

PEMA1	<i>Peromyscus maniculatus</i>
MIPE1	<i>Microtus pennsylvanicus</i>

**Table E 4.2-5. Terrestrial Arthropod Tissue Samples Exceeding Background at OU1 Sites**

Site	Species	Analyte	OU1 Sample	RFP Background	Units
MA04A	ACRI	Cadmium	6.4	3.6	mg/kg
MA04A	ACRI	Copper	204	66.3	mg/kg
MA04A	ACRI	Zinc	210	144	mg/kg
MD01A	ACRI	Cadmium	3.8	3.6	mg/kg
MD01A	ACRI	Copper	133	73.9	mg/kg
MD01A	ACRI	Zinc	144	138	mg/kg
MD01A	ACRI	Plutonium 239/240	0.027	0	pCi/g
MG03A	ACRI	Cadmium	4.6	3.6	mg/kg
MG03A	ACRI	Copper	130	73.9	mg/kg
MG03A	ACRI	Lead	3.5	0	mg/kg
MG03A	ACRI	Zinc	242	138	mg/kg
MG03A	ACRI	Americium 241	0.021	0	pCi/g
MG03A	ACRI	Plutonium 239/240	0.033	0	pCi/g
MR03A	ACRI	Copper	89.2	73.9	mg/kg
MR03A	ACRI	Lead	1.9	0	mg/kg
MR03A	ACRI	Zinc	162	138	mg/kg

Key	
ACRI	aggregate Orthoptera

Table E 4.2-6. Fish Tissue Samples Exceeding Background at OU1 Sites

Site	Species	Analyte	OU1 Sample	RFP Background	Units
SW039	NONE				
SW033	SEAT1	Lead	5.7	4.3	mg/kg
SW033	SEAT1	Total Uranium	0.25	0	mg/kg
SW032	NONE				
SWC001	NONE				
SWC001	LECY1	Copper	52.6	10.3	mg/kg
SWC001	LECY1	Lead	2.6	1.9	mg/kg
SWC001	MISA1	Chromium	4.5	3.8	mg/kg
SWC001	NOCR1	Cadmium	3.3	0	mg/kg
SWC001	NOCR1	Cadmium	3.9	0	mg/kg
SWC001	NOCR1	Chromium	10.4	7.4	mg/kg
SWC001	NOCR1	Chromium	8.9	7.4	mg/kg
SWC001	NOCR1	Copper	103	40.9	mg/kg
SWC001	NOCR1	Copper	73	40.9	mg/kg
SWC001	SEAT1	Copper	657	40.9	mg/kg
SWC001	SEAT1	Lead	6.4	4.3	mg/kg
WOR103	SEAT1	Copper	65.7	40.9	mg/kg
WOR103	SEAT1	Copper	58.4	40.9	mg/kg
SW026	SEAT1	Lead	5.8	4.3	mg/kg
SW026	SEAT1	Radium 226	24	0	pCi/g
SW026	SEAT1	Total Uranium	0.24	0	pCi/g
SW026	SEAT1	Total Uranium	0.28	0	pCi/g
WOR101	NOCR1	Copper	76.9	40.9	mg/kg
WOR101	NOCR1	Copper	83.9	40.9	mg/kg
WOR101	PIPR1	Plutonium 239/240	0.034	0	pCi/g
WOPO02	CACO1	Copper	115	40.9	mg/kg
WOPO02	CACO1	Lead	4.4	4.3	mg/kg
WOPO02	SEAT1	Total Uranium	0.31	0	pCi/g
WOPO02	SEAT1	Total Uranium	0.38	0	pCi/g
WOPO02	SEAT1	Total Uranium	0.21	0	pCi/g
WOPO02	SEAT1	Total Uranium	0.29	0	pCi/g

**Key**

SEAT1	<i>Semotilus atromaculatus</i>
LECY1	<i>Lepomis cyanellus</i>
MISA1	<i>Micropterus salmonoides</i>
NOCR1	<i>Notropis cornutus</i>
PIPR1	<i>Pimephales promelas</i>
CACO1	<i>Catostomus commersoni</i>

Table E 4.3-1. Distribution of Contaminants of Concern Among Environmental Media at OU1

Chemical	Maximum Concentration <sup>1</sup>	Background
<b>Soils</b>		
chromium	80.5	18.3
lead	228.0	40.0
zinc	182.0	71.2
<b>Surface Water<sup>2</sup></b>		
chromium	63.0	10.0
lead	38.4	4.0
mercury	1.0	0.3
<b>Sediments</b>		
None above background that pose ecological risk		

<sup>1</sup> Maximum concentration in mg/kg during sampling program.

<sup>2</sup> Dissolved metal in  $\mu\text{g/l}$ .

Table E 4.3-2. Hazard Quotient Values for Chromium in OU1 IHSS Source Areas

LOCATION	CHEMICAL	RESULT	UNITS	HQ VEGETATION	RISK	HQ INVERTEBRATES	RISK
<b>OU1 IHSS AREAS</b>							
RA013	CHROMIUM	18.70	MG/KG	1.02	L	1.02	L
RA014	CHROMIUM	9.80	MG/KG	0.54	L	0.54	L
RA016	CHROMIUM	13.40	MG/KG	0.73	L	0.73	L
RA017	CHROMIUM	8.40	MG/KG	0.46	L	0.46	L
RA018	CHROMIUM	64.20	MG/KG	3.51	M	3.51	M
RA029	CHROMIUM	9.40	MG/KG	0.51	L	0.51	L
RA030	CHROMIUM	10.00	MG/KG	0.55	L	0.55	L
RA031	CHROMIUM	80.50	MG/KG	4.4	M	4.4	M
RA032	CHROMIUM	21.60	MG/KG	1.18	L	1.18	L
RA032	CHROMIUM	27.30	MG/KG	1.49	L	1.49	L
RA033	CHROMIUM	22.00	MG/KG	1.2	L	1.2	L
RA035	CHROMIUM	15.90	MG/KG	0.87	L	0.87	L
RA036	CHROMIUM	16.60	MG/KG	0.91	L	0.91	L
RA037	CHROMIUM	21.60	MG/KG	1.18	L	1.18	L
Mean=		24.24		1.33		1.33	
Std deviation=		21.392		1.169		1.169	
Range=		8.4-80.5		0.46-4.4		0.46-4.4	

**NON-IHSS AREAS**

RA010	CHROMIUM	14.70	MG/KG	0.8	L	0.8	L
RA012	CHROMIUM	14.60	MG/KG	0.8	L	0.8	L
RA015	CHROMIUM	23.30	MG/KG	1.27	L	1.27	L
RA019	CHROMIUM	12.80	MG/KG	0.7	L	0.7	L
RA020	CHROMIUM	14.60	MG/KG	0.8	L	0.8	L
RA021	CHROMIUM	13.60	MG/KG	0.74	L	0.74	L
RA022	CHROMIUM	15.50	MG/KG	0.85	L	0.85	L
RA023	CHROMIUM	14.90	MG/KG	0.81	L	0.81	L
RA024	CHROMIUM	13.50	MG/KG	0.74	L	0.74	L
RA025	CHROMIUM	11.10	MG/KG	0.61	L	0.61	L
RA026	CHROMIUM	13.50	MG/KG	0.74	L	0.74	L
RA027	CHROMIUM	17.40	MG/KG	0.95	L	0.95	L
RA028	CHROMIUM	17.30	MG/KG	0.95	L	0.95	L
Mean=		15.14		0.83		0.83	
Std deviation=		2.984		0.162		0.162	
Range=		11.1-23.3		0.61-1.27		0.61-1.27	

**ROCK CREEK AREAS**

RA001	CHROMIUM	20.20	MG/KG	1.1	L	1.1	L
RA002	CHROMIUM	21.60	MG/KG	1.18	L	1.18	L
RA003	CHROMIUM	16.30	MG/KG	0.89	L	0.89	L
RA004	CHROMIUM	22.00	MG/KG	1.2	L	1.2	L
RA005	CHROMIUM	15.00	MG/KG	0.82	L	0.82	L
RA006	CHROMIUM	14.90	MG/KG	0.81	L	0.81	L
RA007	CHROMIUM	15.50	MG/KG	0.85	L	0.85	L
RA008	CHROMIUM	12.80	MG/KG	0.7	L	0.7	L
RA009	CHROMIUM	17.10	MG/KG	0.93	L	0.93	L
Mean=		17.27		0.87		0.87	
Std deviation=		3.251		0.271		0.271	
Range=		12.8-22.0		0.7-1.18		0.7-1.18	

Table E 4.3-3. Hazard Quotient Values for Zinc in OU1 IHSS Source Areas

LOCATION	CHEMICAL	RESULT	UNITS	HQ VEGETATION	RISK	HQ INVERTEBRATES	RISK
OU1 IHSS AREAS							
RA013	ZINC	73.00	MG/KG	1.03	L	0.07	L
RA014	ZINC	182.00	MG/KG	2.56	L	0.18	L
RA016	ZINC	60.60	MG/KG	0.85	L	0.06	L
RA017	ZINC	64.60	MG/KG	0.91	L	0.06	L
RA018	ZINC	55.30	MG/KG	0.78	L	0.06	L
RA029	ZINC	56.90	MG/KG	0.8	L	0.06	L
RA030	ZINC	59.50	MG/KG	0.84	L	0.06	L
RA031	ZINC	165.00	MG/KG	2.32	L	0.17	L
RA032	ZINC	96.20	MG/KG	1.35	L	0.1	L
RA033	ZINC	66.20	MG/KG	0.93	L	0.07	L
RA035	ZINC	79.20	MG/KG	1.11	L	0.08	L
RA036	ZINC	130.00	MG/KG	1.83	L	0.13	L
RA037	ZINC	96.00	MG/KG	1.35	L	0.1	L
Mean Concentration=		86.89		1.22		0.09	
Std deviation=		36.15		0.508		0.037	
Range=		55.3-182		0.8-2.56		0.06-0.18	

## NON-IHSS AREAS

RA010	ZINC	66.80	MG/KG	0.94	L	0.07	L
RA012	ZINC	61.30	MG/KG	0.86	L	0.06	L
RA015	ZINC	59.60	MG/KG	0.84	L	0.06	L
RA015	ZINC	72.00	MG/KG	1.01	L	0.07	L
RA019	ZINC	55.90	MG/KG	0.79	L	0.06	L
RA020	ZINC	66.10	MG/KG	0.93	L	0.07	L
RA021	ZINC	52.90	MG/KG	0.74	L	0.05	L
RA022	ZINC	65.50	MG/KG	0.92	L	0.07	L
RA023	ZINC	56.10	MG/KG	0.79	L	0.06	L
RA024	ZINC	62.80	MG/KG	0.88	L	0.06	L
RA025	ZINC	58.20	MG/KG	0.82	L	0.06	L
RA026	ZINC	64.40	MG/KG	0.9	L	0.06	L
RA027	ZINC	61.50	MG/KG	0.86	L	0.06	L
RA028	ZINC	58.50	MG/KG	0.82	L	0.06	L
Mean Concentration=		60.19		0.85		0.06	
Std deviation=		4.53		0.063		0.006	
Range=		52.9-72		0.74-1.01		0.05-0.07	

## ROCK CREEK AREAS

RA001	ZINC	61.70	MG/KG	0.87	L	0.06	L
RA002	ZINC	48.70	MG/KG	0.68	L	0.05	L
RA002	ZINC	90.20	MG/KG	1.27	L	0.09	L
RA003	ZINC	62.80	MG/KG	0.88	L	0.06	L
RA004	ZINC	61.80	MG/KG	0.87	L	0.06	L
RA004	ZINC	78.00	MG/KG	1.1	L	0.08	L
RA005	ZINC	56.50	MG/KG	0.79	L	0.06	L
RA006	ZINC	56.00	MG/KG	0.79	L	0.06	L
RA007	ZINC	57.10	MG/KG	0.8	L	0.06	L
RA008	ZINC	50.80	MG/KG	0.71	L	0.05	L
RA009	ZINC	57.90	MG/KG	0.81	L	0.06	L
Mean Concentration=		61.98		0.87		0.06	
Std deviation=		12.74		0.181		0.013	
Range=		48.7-90.2		0.68-1.27		0.05-0.09	



Table E 4.3-4. Hazard Quotient Values for Lead in OU1 IHSS Source Areas

LOCATION	CHEMICAL	RESULT	UNITS	HQ VEGETATION	RISK	HQ INVERTEBRATES	RISK
<b>OU1 IHSS AREAS</b>							
RA013	LEAD	19.30	MG/KG	0.39	L	0.48	L
RA014	LEAD	20.20	MG/KG	0.4	L	0.5	L
RA016	LEAD	26.10	MG/KG	0.52	L	0.65	L
RA017	LEAD	24.40	MG/KG	0.49	L	0.61	L
RA018	LEAD	34.40	MG/KG	0.69	L	0.86	L
RA029	LEAD	28.10	MG/KG	0.56	L	0.7	L
RA030	LEAD	7.10	MG/KG	0.14	L	0.18	L
RA031	LEAD	46.50	MG/KG	0.93	L	1.16	L
RA032	LEAD	228.00	MG/KG	4.56	M	5.69	M
RA033	LEAD	47.40	MG/KG	0.95	L	1.18	L
RA035	LEAD	23.00	MG/KG	0.46	L	0.57	L
RA036	LEAD	15.90	MG/KG	0.32	L	0.4	L
RA037	LEAD	30.10	MG/KG	0.6	L	0.75	L

Mean= 42.35                      0.85                      1.06  
 Std deviation= 56.899                      1.138                      1.42  
 Range= 7.1-228                      0.14-4.56                      0.18-5.69

**NON-IHSS AREAS**

RA012	LEAD	41.70	MG/KG	0.83	L	1.04	L
RA015	LEAD	39.70	MG/KG	0.79	L	0.99	L
RA019	LEAD	36.20	MG/KG	0.72	L	0.9	L
RA020	LEAD	32.00	MG/KG	0.64	L	0.8	L
RA021	LEAD	24.20	MG/KG	0.48	L	0.6	L
RA022	LEAD	45.30	MG/KG	0.91	L	1.13	L
RA023	LEAD	37.50	MG/KG	0.75	L	0.94	L
RA024	LEAD	36.60	MG/KG	0.73	L	0.91	L
RA025	LEAD	33.80	MG/KG	0.68	L	0.84	L
RA026	LEAD	20.70	MG/KG	0.41	L	0.52	L
RA027	LEAD	30.70	MG/KG	0.61	L	0.77	L
RA028	LEAD	36.20	MG/KG	0.72	L	0.9	L

Mean= 34.55                      0.69                      0.86  
 Std deviation= 6.957                      0.14                      0.173  
 Range= 20.7-45.3                      0.41-0.91                      0.52-1.13

**ROCK CREEK AREAS**

RA001	LEAD	34.50	MG/KG	0.69	L	0.86	L
RA002	LEAD	44.00	MG/KG	0.88	L	1.1	L
RA003	LEAD	30.60	MG/KG	0.61	L	0.76	L
RA004	LEAD	42.30	MG/KG	0.85	L	1.06	L
RA005	LEAD	34.90	MG/KG	0.7	L	0.87	L
RA006	LEAD	35.10	MG/KG	0.7	L	0.88	L
RA007	LEAD	36.80	MG/KG	0.74	L	0.92	L
RA008	LEAD	39.70	MG/KG	0.79	L	0.99	L
RA009	LEAD	37.30	MG/KG	0.75	L	0.93	L

Mean= 37.24                      0.75                      0.93  
 Std deviation= 4.171                      0.084                      0.106  
 Range= 30.6-44                      0.61-0.88                      0.76-1.1

Table E 4.4-1 Terrestrial and Aquatic Taxonomic Richness Percentage Comparison for OU1 EE

Terrestrial Taxonomic Richness			
	OU1	%	Rock Creek %
Vascular Plants	219	54.9	247 57.7
Arthropods (1)	103	25.8	110 25.7
Amphibians	1	0.3	2 0.5
Reptiles	5	1.3	4 0.9
Birds	56	14.0	50 11.7
Mammals	15	3.8	15 3.5
<b>TOTAL NUMBER OF TAXA</b>	<b>399</b>	<b>100</b>	<b>428 100</b>

Aquatic Taxonomic Richness			
	OU1	%	Rock Creek %
Plankton (2)	86	43.9	94 55.3
Benthic Macroinvertebrates	102	52.0	73 42.9
Amphibians (3)	1	0.5	0 0.0
Fish	7	3.6	3 1.8
<b>TOTAL NUMBER OF TAXA</b>	<b>196</b>	<b>100</b>	<b>170 100</b>

(1) Arthropods were identified only to family, all other taxonomic groups were identified to species.

(2) Zooplankton data were not included in this comparison.

(3) This group is comprised only of aquatic larval stage of the tiger salamander.

Table E 4.4-2 Terrestrial and Aquatic Taxonomic Richness Statistical Comparison for OU1 EE

Terrestrial Taxonomic Richness				Chi - Squared Calculation (Ho: OU1 = REF)			
	fo	(1)	OU1	Rock Creek	Summation	fo	$\frac{(fo-fe)^2}{fe}$
Vascular Plants			219	247	466	219	224.8
Arthropods (2)			103	110	213	247	241.2
Amphibians			1	2	3	103	102.8
Reptiles			5	4	9	110	110.2
Birds			56	50	106		
Mammals			15	15	30		
<b>TOTAL NUMBER OF TAXA</b>			<b>399</b>	<b>428</b>	<b>827</b>		
Terrestrial Taxonomic Richness				OU1			
	fo		OU1	Rock Creek	Summation		
Vascular Plants			224.8	241.2	466		
Arthropods			102.8	110.2	213		
Amphibians			1.4	1.6	3		
Reptiles			4.3	4.7	9		
Birds			51.1	54.9	106		
Mammals			14.5	15.5	30		
<b>TOTAL NUMBER OF TAXA</b>			<b>399</b>	<b>428</b>	<b>827</b>		
Terrestrial Taxonomic Richness				OU1			
	fo		OU1	Rock Creek	Summation		
Vascular Plants/OU1			219	247	466		
Vascular Plants/REF			247	110	213		
Arthropods/OU1			103	2	3		
Arthropods/REF			110	4	9		
Amphibians/OU1			1	50	106		
Amphibians/REF			2	15	30		
Reptiles/OU1			5				
Reptiles/REF			4				
Birds/OU1			56				
Birds/REF			50				
Mammals/OU1			15				
Mammals/REF			15				
<b>TOTAL</b>			<b>827</b>	<b>827.00</b>	<b>1.68</b>		
d.f. = 5				Not significant at the 95% conf. int.			

- (1) fo = observed frequency fe = expected frequency  
 (2) Arthropods were identified only to family, all other taxonomic groups were identified to species.  
 (3) Zooplankton data were not included in this comparison.  
 (4) This group is comprised only of aquatic larval stage of the tiger salamander.



Table E 4.4-3 Terrestrial and Aquatic Trophic Level Percentage Comparison for OU1 EE (1)

Terrestrial Trophic Levels (2)	OU1	%	Rock Creek	%
Primary Producers	219	74.0	247	77.7
Herbivores	15	5.1	16	5.0
Omnivores	25	8.4	21	6.6
Predators	32	10.8	25	7.9
Top Carnivores	5	1.7	9	2.8
Total Number of Species	296	100	318	100

**Terrestrial Arthropods only**

Terrestrial Trophic Levels	OU1	%	Rock Creek	%
Detritivores	8	7.8	9	8.2
Herbivores	52	50.5	49	44.5
Omnivores	4	3.9	5	4.5
Predators	24	23.3	27	24.5
Parasites	15	14.6	20	18.2
Total Number of Families	103	100	110	100
TOTAL NUMBER OF TERRESTRIAL TAXA	399		428	

Aquatic Trophic Levels (3)	OU1	%	Rock Creek	%
Detritivores	19	9.7	17	10.0
Primary Producers	86	43.9	94	55.3
Herbivores	23	11.7	24	14.1
Omnivores	41	20.9	13	7.6
Predators	23	11.7	19	11.2
Top Carnivores	2	1.0	1	0.6
Parasites	2	1.0	2	1.2
TOTAL NUMBER OF AQUATIC TAXA	196	100	170	100

(1) Arthropods were identified only to family, all other taxonomic groups were identified to species and were readily comparable.

(2) Terrestrial trophic levels include all vascular plants, reptiles, birds (including raptors), and mammals.

(3) Aquatic trophic groups include all phytoplankton, periphyton, benthic macroinvertebrates, amphibians, and fish.

#### Table E 4.4.4 Terrestrial and Aquatic Trophic Level Statistical Comparison for OU1 EE (1)

Terrestrial Trophic Levels (2)	fo (3)	OU1	Rock Creek	Summation
Primary Producers		219	247	466
Herbivores		15	16	31
Omnivores		25	21	46
Predators		32	25	57
Top Carnivores		5	9	14
TOTAL NUMBER OF SPECIES				
		296	318	614

Terrestrial Trophic Levels	fe	OU1	Rock Creek	Summation
Primary Producers		224.7	241.3	466
Herbivores		14.9	16.1	31
Omnivores		22.2	23.8	46
Predators		27.5	29.5	57
Top Carnivores		6.7	7.3	14
TOTAL NUMBER OF SPECIES				
		296	318	614

(1) Arthropods were identified only to family and therefore, compared separately. All other taxonomic groups were identified to species and were readily comparable.

(2) Terrestrial trophic levels include all vascular plants, reptiles, birds (including raptors), and mammals.

(3) fo = observed frequency    fe = expected frequency

(4) Aquatic trophic groups include all phytoplankton, periphyton, benthic macroinvertebrates, amphibians, and fish.

Chi - Squared Calculation (Ho: OU1 = REF)			
Terrestrial Trophic Levels	fo	(fo-fe) <sup>2</sup>	
		fe	fe
Primary Producer/OU1	219	224.7	0.14
Primary Producer/REF	247	241.3	0.13
Herbivores/OU1	15	14.9	0.00
Herbivores/REF	16	16.1	0.00
Omnivores/OU1	25	22.2	0.36
Omnivores/REF	21	23.8	0.33
Predators/OU1	32	27.5	0.74
Predators/REF	25	29.5	0.69
Top Carnivores/OU1	5	6.7	0.45
Top Carnivores/REF	9	7.3	0.42
<b>Total</b>	<b>614</b>	<b>614.00</b>	<b>3.28</b>

d.f. = 4

Table E 4.4-4 Terrestrial and Aquatic Trophic Level Statistical Comparison for OU1 EE (1)

Terrestrial Arthropods only					Terrestrial Arthropods only				
Terrestrial Trophic Levels					Chi - Squared Calculation (Ho: OU1 = REF)				
	fo	OU1	Rock Creek	Summation	Terrestrial Trophic Levels	fo	fe	$\frac{(fo-fe)^2}{fe}$	
Detritivores		8	9	17	Primary Producer/OU1	8	8.2	0.01	
Herbivores		52	49	101	Primary Producer/REF	9	8.8	0.01	
Omnivores		4	5	9	Herbivores/OU1	52	48.8	0.20	
Predators		24	27	51	Herbivores/REF	49	52.2	0.19	
Parasites		15	20	35	Omnivores/OU1	4	4.4	0.03	
					Omnivores/REF	5	4.6	0.03	
TOTAL NUMBER OF FAMILIES		103	110	213	Predators/OU1	24	24.7	0.02	
TOTAL NUMBER OF TERRESTRIAL TA		399	428		Predators/REF	27	26.3	0.02	
					Top Carnivores/OU1	15	16.9	0.22	
Terrestrial Arthropods only					Top Carnivores/REF	20	18.1	0.20	
Terrestrial Trophic Levels					Total	213	213	0.92	
Detritivores	fe	OU1	Rock Creek	Summation	d.f. = 4	Not significant at the 95% conf. int.			
Herbivores		8.2	8.8	17					
Omnivores		48.8	52.2	101					
Predators		4.4	4.6	9					
Parasites		24.7	26.3	51					
		16.9	18.1	35					
TOTAL NUMBER OF FAMILIES		103	110	213					

(1) Arthropods were identified only to family and therefore, compared separately. All other taxonomic groups were identified to species and were readily comparable.

(2) Terrestrial trophic levels include all vascular plants, reptiles, birds (including raptors), and mammals.

(3) fo = observed frequency fe = expected frequency

(4) Aquatic trophic groups include all phytoplankton, periphyton, benthic macroinvertebrates, amphibians, and fish.

Table E 4.4-4 Terrestrial and Aquatic Trophic Level Statistical Comparison for OU1 EE (1)

Aquatic Trophic Levels	fo	OU1	Rock Creek	Summation	Chi - Squared Calculation (Ho: OU1 = REF)			
					fo	fe	fe	$(fo-fe)^2/fe$
Detritivores		19	17	36	19	19.3	19.3	0.00
Primary Producers		86	94	180	17	16.7	16.7	0.00
Herbivores		23	24	47	86	96.4	96.4	1.12
Omnivores		41	13	54	94	83.6	83.6	1.29
Predators		23	19	42	23	25.2	25.2	0.19
Top Carnivores		2	1	3	24	21.8	21.8	0.22
Parasites		2	2	4	41	28.9	28.9	5.05
<b>TOTAL NUMBER OF AQUATIC TAXA</b>		<b>196</b>	<b>170</b>	<b>366</b>	<b>13</b>	<b>25.1</b>	<b>25.1</b>	<b>5.82</b>
					<b>23</b>	<b>22.5</b>	<b>22.5</b>	<b>0.01</b>
Aquatic Trophic Levels	fe	OU1	Rock Creek	Summation				
Detritivores		19.3	16.7	36	19	19.5	19.5	0.01
Primary Producers		96.4	83.6	180	2	1.6	1.6	0.10
Herbivores		25.2	21.8	47	1	1.4	1.4	0.11
Omnivores		28.9	25.1	54	2	2.1	2.1	0.01
Predators		22.5	19.5	42	2	1.9	1.9	0.01
Top Carnivores		1.6	1.4	3				
Parasites		2.1	1.9	4				
<b>TOTAL NUMBER OF AQUATIC TAXA</b>		<b>196</b>	<b>170</b>	<b>366</b>	<b>366</b>	<b>366</b>	<b>366</b>	<b>13.94</b>
					d.f. = 6			
					Significant at the 95% conf. int.			

(1) Arthropods were identified only to family and therefore, compared separately. All other taxonomic groups were identified to species and were readily comparable.

(2) Terrestrial trophic levels include all vascular plants, reptiles, birds (including raptors), and mammals.

(3) fo = observed frequency fe = expected frequency

(4) Aquatic trophic groups include all phytoplankton, periphyton, benthic macroinvertebrates, amphibians, and fish.



Table E 4.4-5 Results for Plant and Animal Endpoints for OU1 EE

Endpoints	Habitat Types Compared							
	Xeric Grassland		Mesic Grassland		Woodland		Marshland	
	OU1	Rock Creek	OU1	Rock Creek	OU1	Rock Creek	OU1	Rock Creek
<b>PLANTS</b>								
<b>TREE AND SHRUB DENSITY</b>								
Mean	0	5	0	24	474	237	4	31
Standard Deviation	0.0	14.0	0.0	58.5	185.6	117.5	13.7	35.6
Standard Error	0.0	4.4	0.0	18.5	47.9	37.1	3.5	11.3
<b>SPECIES RICHNESS</b>								
Mean	23	51	45	44	51	62	18	35
Standard Deviation	2.2	6.3	4.7	8.9	21.9	9.0	7.6	9.5
Standard Error	0.6	2.0	1.2	2.8	5.7	2.8	2.0	3.0
<b>STANDING CROP</b>								
Mean	1304	1232	1901	1699	1054	774	3378	2410
Standard Deviation	903.5	322.2	1065.1	666.3	955.8	626.0	1847.6	1138.9
Standard Error	165.0	72.0	194.5	121.7	174.5	114.3	337.3	207.9
<b>ANIMALS</b>								
<b>TERRESTRIAL ARTHROPOD - SUMMER</b>								
<b>NUMBER OF INDIVIDUALS</b>								
Mean	35	245	194	292	370	728	223	355
Standard Deviation	50.0	33.6	54.3	71.3	36.6	64.0	95.8	52.2
Standard Error	28.8	19.4	31.4	41.2	21.1	36.9	55.3	30.1
<b>SMALL MAMMALS - SPRING</b>								
<b>NUMBER OF INDIVIDUALS</b>								
Mean	0	6	6	7	15	37	39	11
Standard Deviation	0.0	0.8	3.4	2.4	6.8	5.2	16.0	5.8
Standard Error	0.0	0.4	1.7	1.2	3.4	2.6	8.2	2.9
<b>SPECIES RICHNESS</b>								
Mean	0	1	2	2	2	3	3	2
Standard Deviation	0.0	0.0	1.2	0.4	0.0	0.5	0.4	0.0
Standard Error	0.0	0.0	0.6	0.2	0.0	0.3	0.2	0.0
<b>LIVE WEIGHTS OF DEER MICE - ADULT MALE</b>								
Mean	0	19	20	23	20	21	21	21
Standard Deviation	0.0	1.6	2.3	1.8	1.3	0.1	1.1	1.9
Standard Error	0.0	0.9	1.4	0.9	0.7	0.1	0.5	1.1
<b>LIVE WEIGHTS OF DEER MICE - ADULT FEMALE</b>								
Mean	0	23	21	23	20	22	19	23
Standard Deviation	0.0	0.4	1.7	2.0	0.9	1.3	0.9	1.8
Standard Error	0.0	0.2	1.2	1.0	0.5	0.7	0.5	1.1

**Units for Measurements:**

Shrub and tree density in stems per 100 square meters

Vegetation species richness in number of species

Standing crop in kilogram per hectare

Number of individuals of terrestrial arthropods per three sweep net sessions.

Number of small mammals per 100 trap nights with no repeats

Small mammal species richness (number of species) per 100 trap nights

Live weights of small mammals in grams

Table E 4.4-5 Results for Plant and Animal Endpoints for OU1 EE

Endpoints	Habitat Types Compared							
	Xeric Grassland		Mesic Grassland		Woodland		Marshland	
	OU1	Rock Creek	OU1	Rock Creek	OU1	Rock Creek	OU1	Rock Creek
<b>SMALL MAMMALS (CONT.)</b>								
<b>LIVE WEIGHTS OF MEADOW VOLES - ADULT MALE</b>								
Mean	0	0	44	37	37	46	51	45
Standard Deviation	0.0	0.0	0.0	0.0	4.6	3.1	5.0	10.8
Standard Error	0.0	0.0	0.0	0.0	2.3	1.8	2.5	5.4
<b>LIVE WEIGHTS OF MEADOW VOLES - ADULT FEMALE</b>								
Mean	0	0	35	38	45	45	46	50
Standard Deviation	0.0	0.0	0.0	8.0	6.4	6.3	2.9	11.6
Standard Error	0.0	0.0	0.0	5.7	3.7	3.1	1.5	6.7
<b>SMALL MAMMALS - FALL</b>								
<b>NUMBER OF INDIVIDUALS</b>								
Mean	1	13	10	20	25	43	31	41
Standard Deviation	0.0	8.7	3.3	2.9	13.0	3.6	13.0	6.2
Standard Error	0.0	4.3	1.7	1.4	6.4	1.8	6.3	3.1
<b>SPECIES RICHNESS</b>								
Mean	1	2	2	2	3	2	2	2
Standard Deviation	0.4	0.9	0.7	0.5	0.5	0.0	0.4	0.0
Standard Error	0.4	0.4	0.4	0.3	0.3	0.0	0.2	0.0
<b>LIVE WEIGHTS OF DEER MICE - ADULT MALE</b>								
Mean	0	19	18	19	19	19	22	19
Standard Deviation	0.0	1.4	1.7	1.4	0.7	0.6	2.0	0.7
Standard Error	0.0	0.7	1.0	0.7	0.4	0.3	1.0	0.4
<b>LIVE WEIGHTS OF DEER MICE - ADULT FEMALE</b>								
Mean	0	16	22	18	22	18	21	19
Standard Deviation	0.0	2.0	2.4	1.0	1.6	1.0	0.4	2.5
Standard Error	0.0	1.2	1.2	0.5	0.8	0.5	0.2	1.3
<b>LIVE WEIGHTS OF MEADOW VOLES - ADULT MALE</b>								
Mean	0	35	41	40	50	35	46	38
Standard Deviation	0.0	0.0	2.1	0.0	4.4	7.3	3.5	8.9
Standard Error	0.0	0.0	1.2	0.0	2.2	3.6	1.8	4.4
<b>LIVE WEIGHTS OF MEADOW VOLES - ADULT FEMALE</b>								
Mean	0	38	42	35	47	31	50	38
Standard Deviation	0.0	0.0	2.4	0.0	6.8	5.8	4.1	1.0
Standard Error	0.0	0.0	1.7	0.0	3.4	3.3	2.1	0.5

**Units for Measurements:**

Shrub and tree density in stems per 100 square meters

Vegetation species richness in number of species

Standing crop in kilogram per hectare

Number of individuals of terrestrial arthropods per three sweep net sessions.

Number of small mammals per 100 trap nights with no repeats

Small mammal species richness (number of species) per 100 trap nights

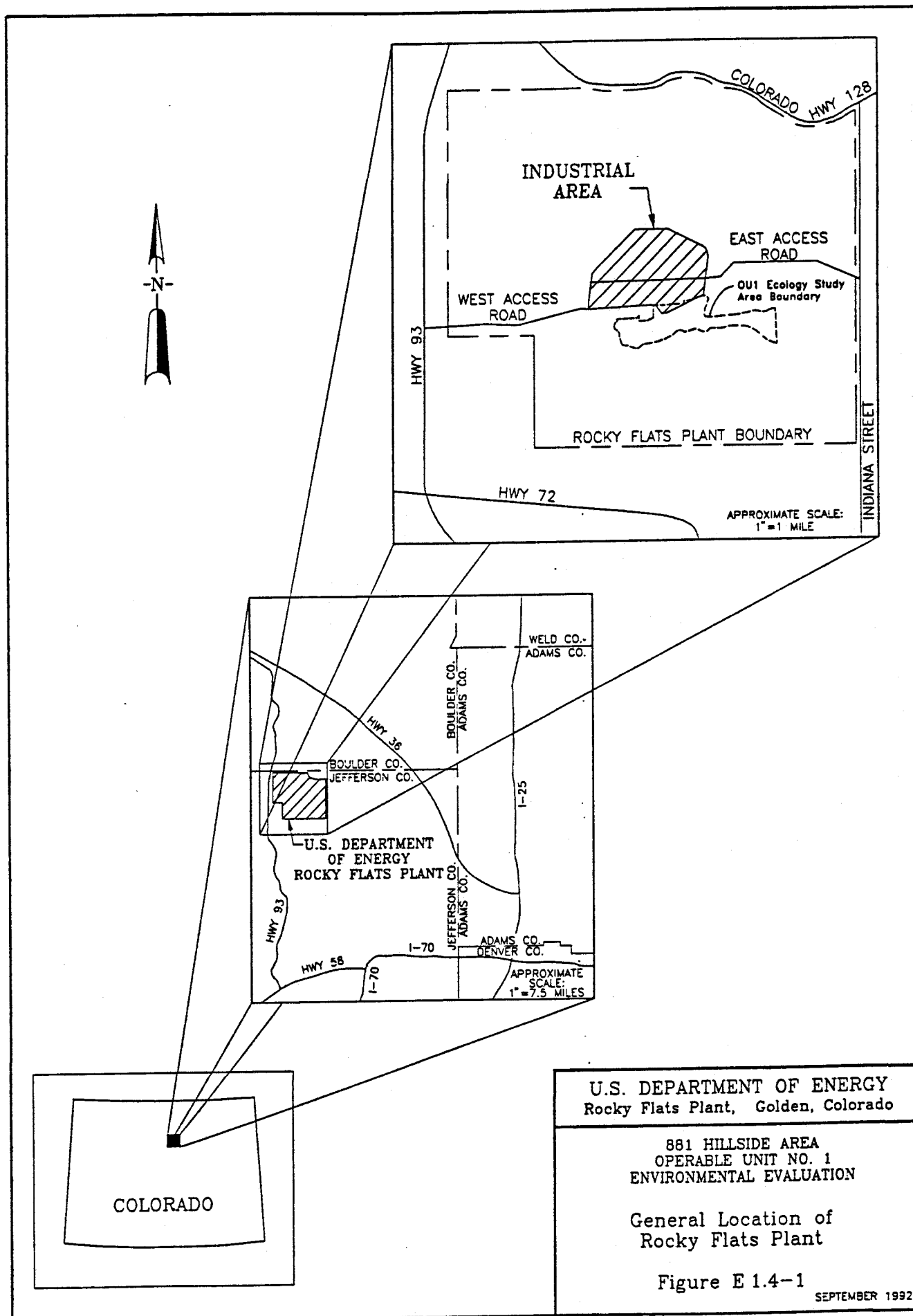
Live weights of small mammals in grams

## Appendix E • Figures

Environmental  
Evaluation

Phase III  
RFI/RI Report

U1 OUI OU  
881 Hillside  
OUI OUI O  
Hillside 881  
1 OUI OUI  
side 881 Hill  
U1 OUI OU  
881 Hillside  
OUI OUI O  
Hillside 881



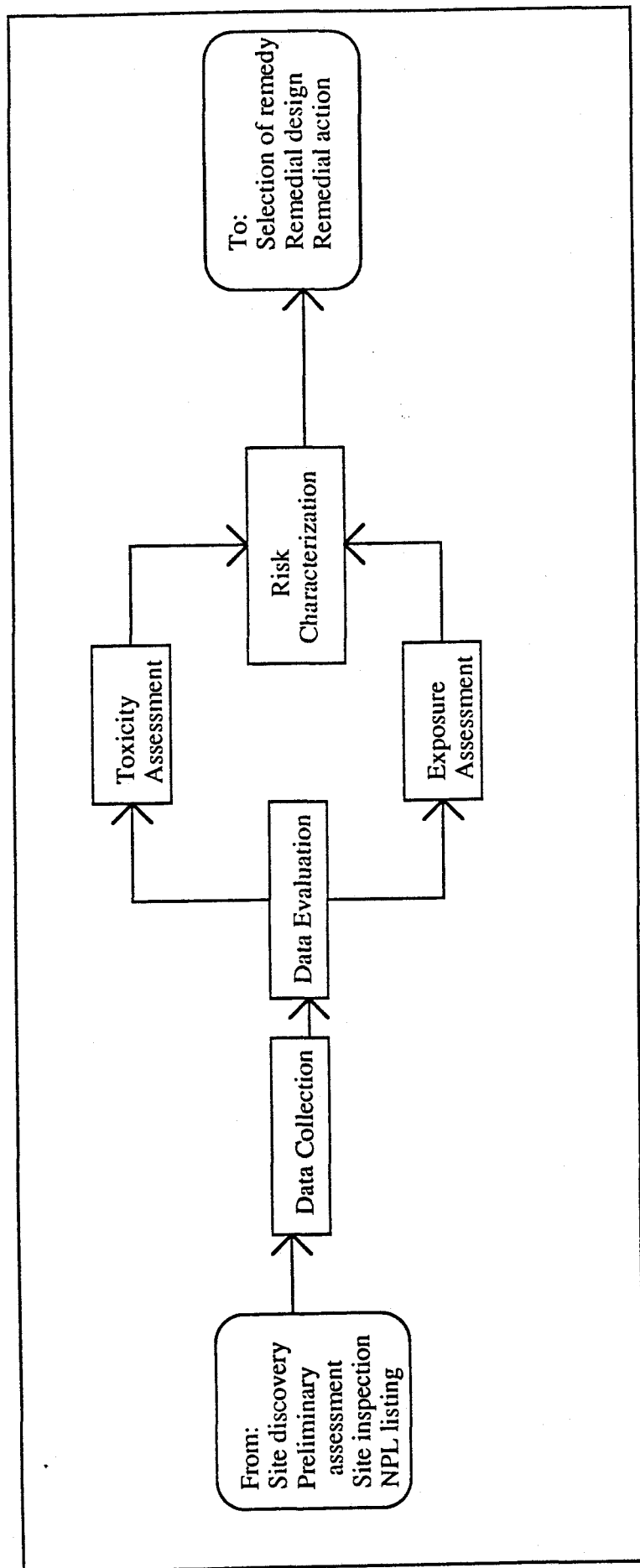
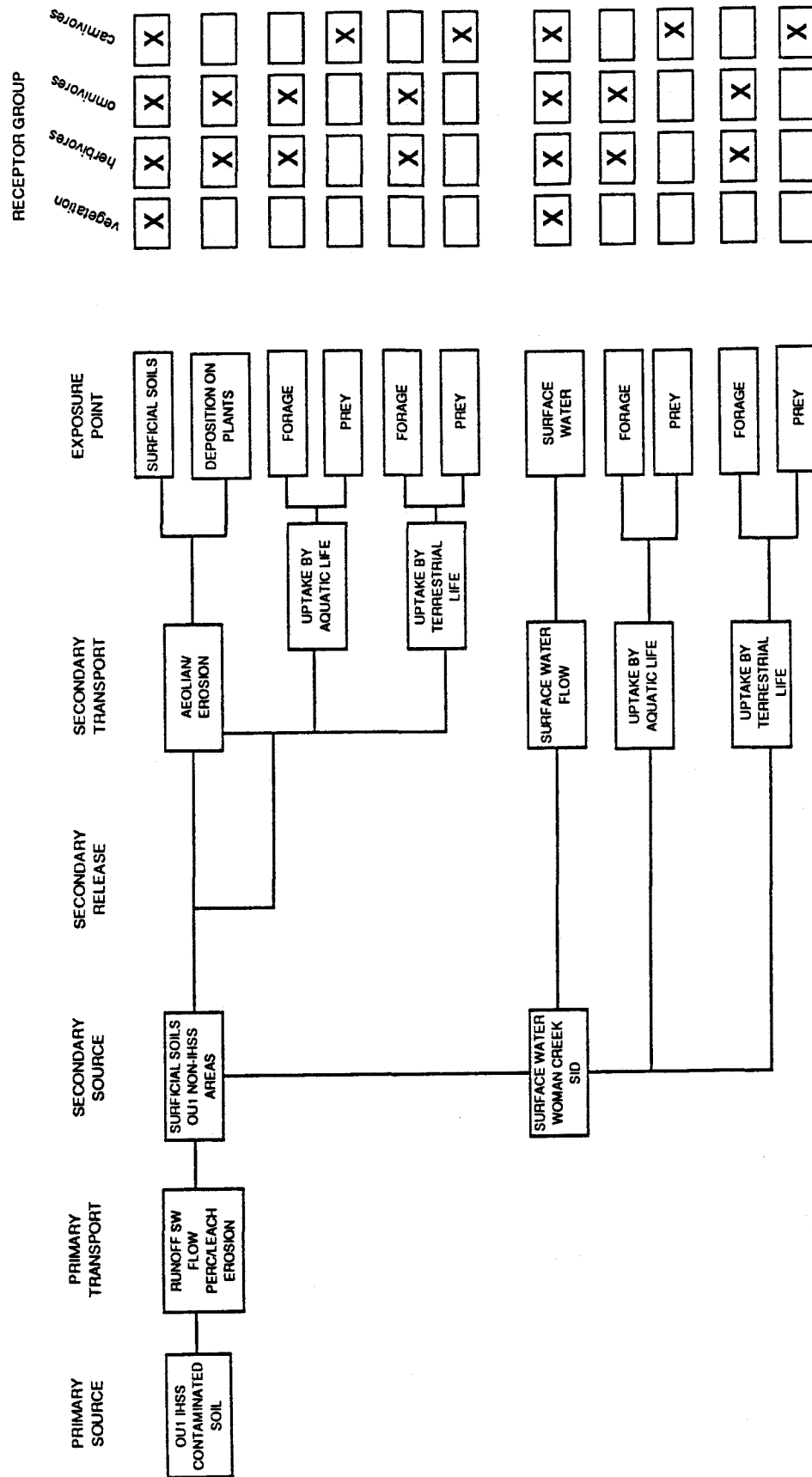


Figure E 3.1-1. Components of Ecological Risk Assessment at OU1



KEY ASSUMPTIONS:  
 1. COC's are primarily metals  
 2. Addition of vols or semivols will cause addition of volatilization pathways

Figure E 3.2-1 Conceptual Exposure Model for OU1 Ecological Risk Assessment

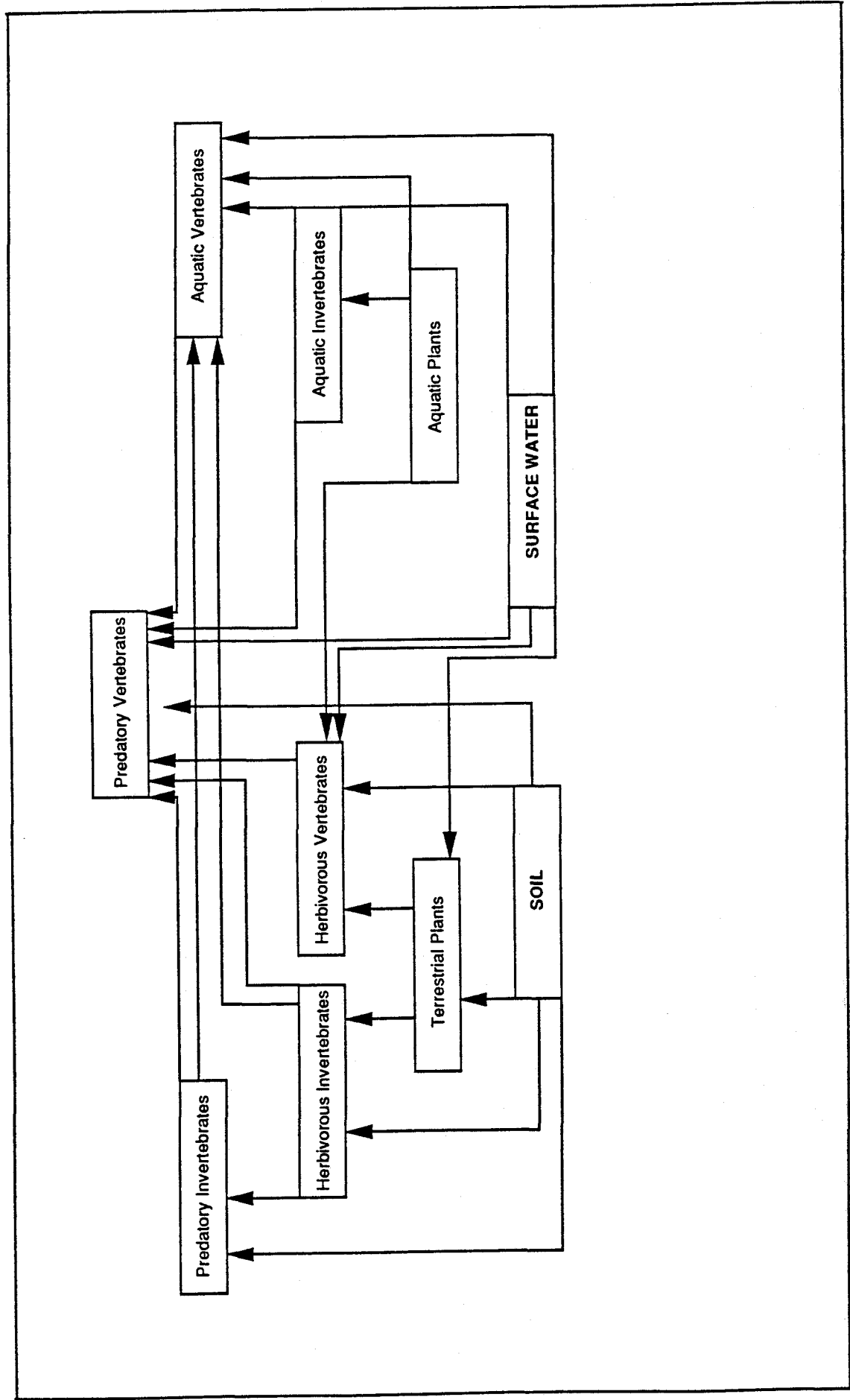
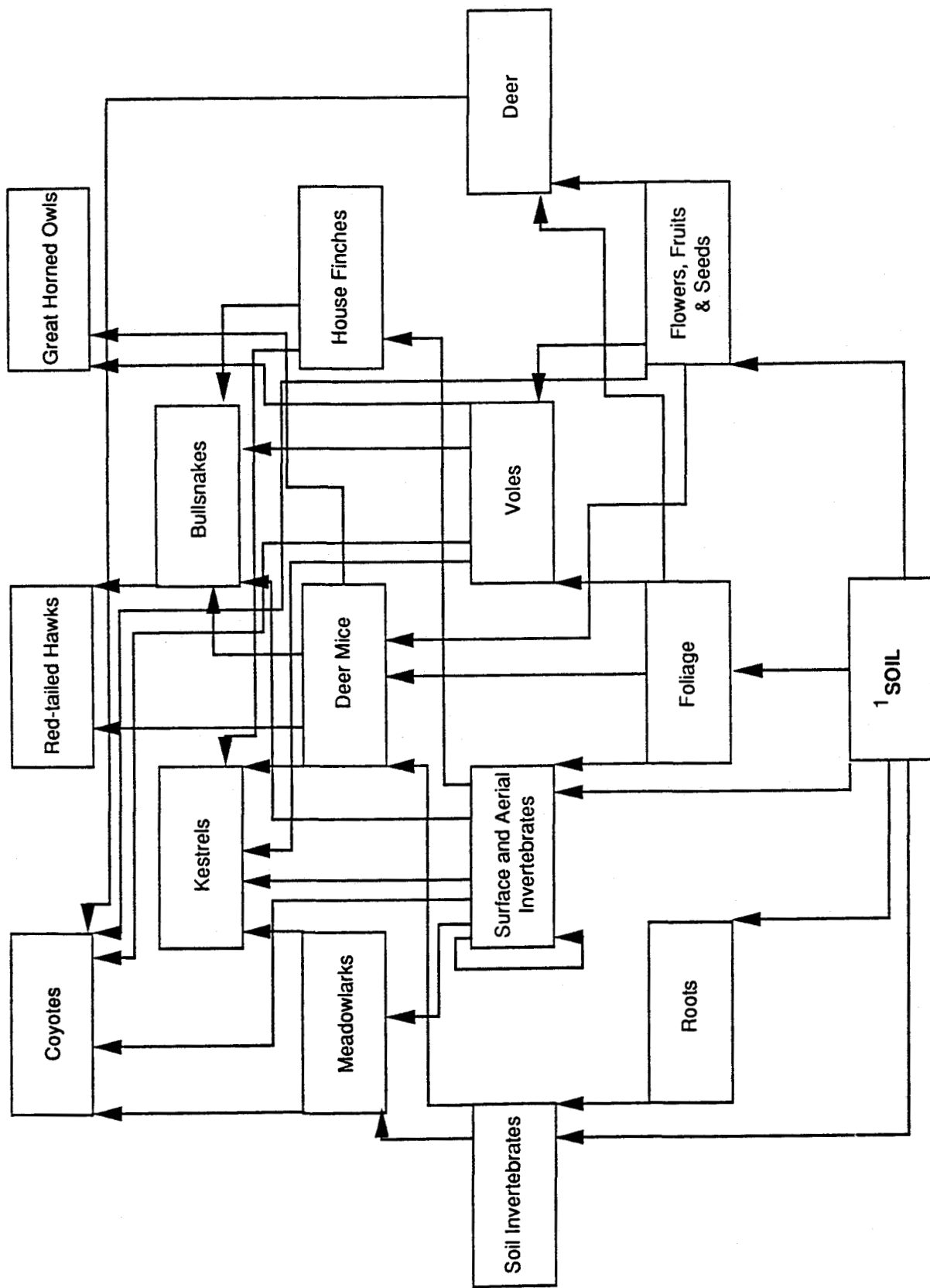


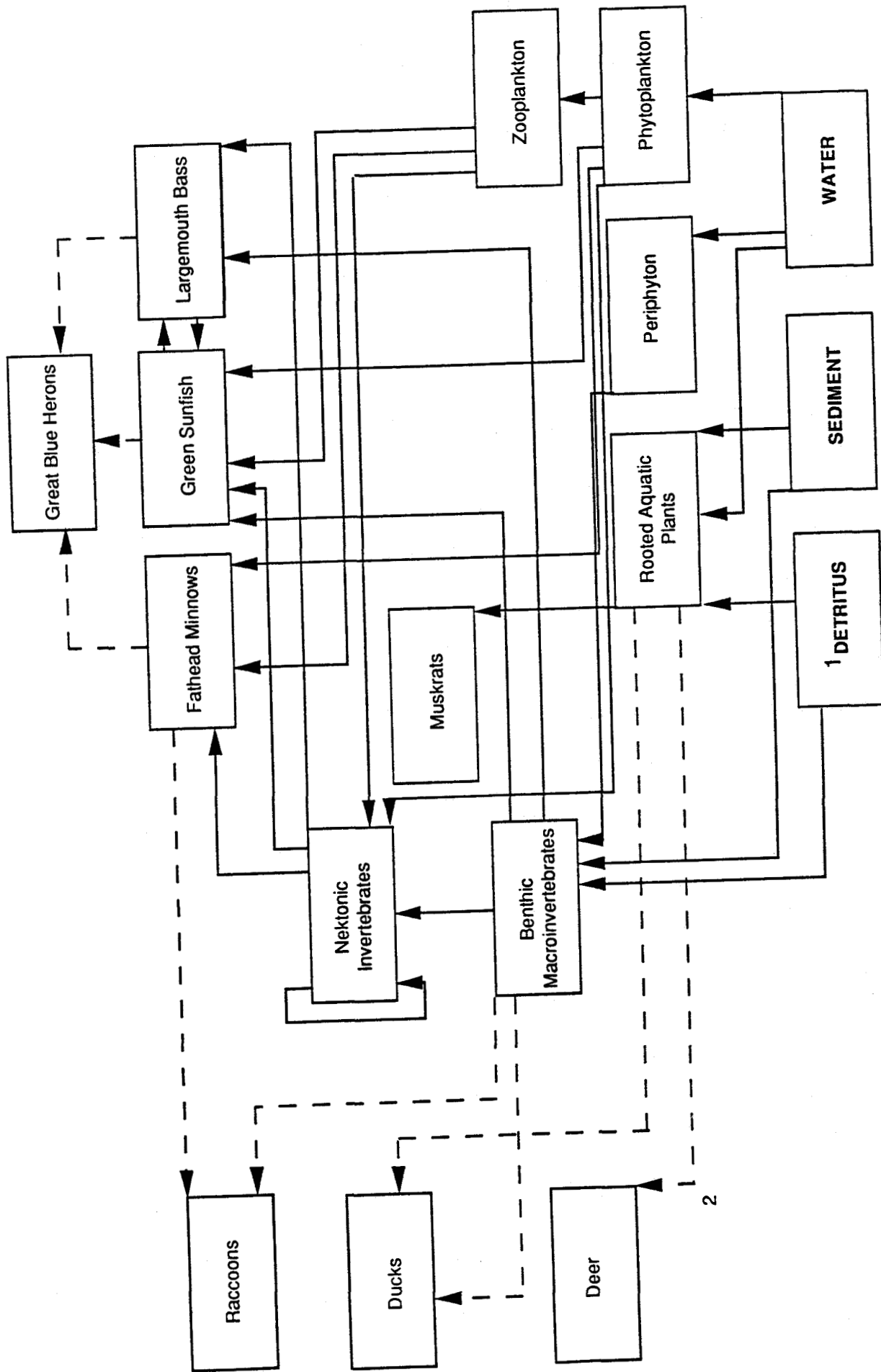
Figure E 3.2-2. Generalized Food Web and Exposure Pathway of a Terrestrial Ecosystem with an Associated Aquatic Ecosystem



<sup>1</sup> All trophic groups may be directly exposed to contaminants in soil and/or surface water (see text).

Figure E 3.2-3. Detailed Terrestrial Food Web, OU 1





<sup>1</sup> All trophic groups may be directly exposed to contaminants in water, sediment, or detritus (see text).

<sup>2</sup> Dashed lines indicate pathways for aquatic to terrestrial food webs.

Figure E 3.2-4. Detailed Aquatic Food Web, OU 1 1,2

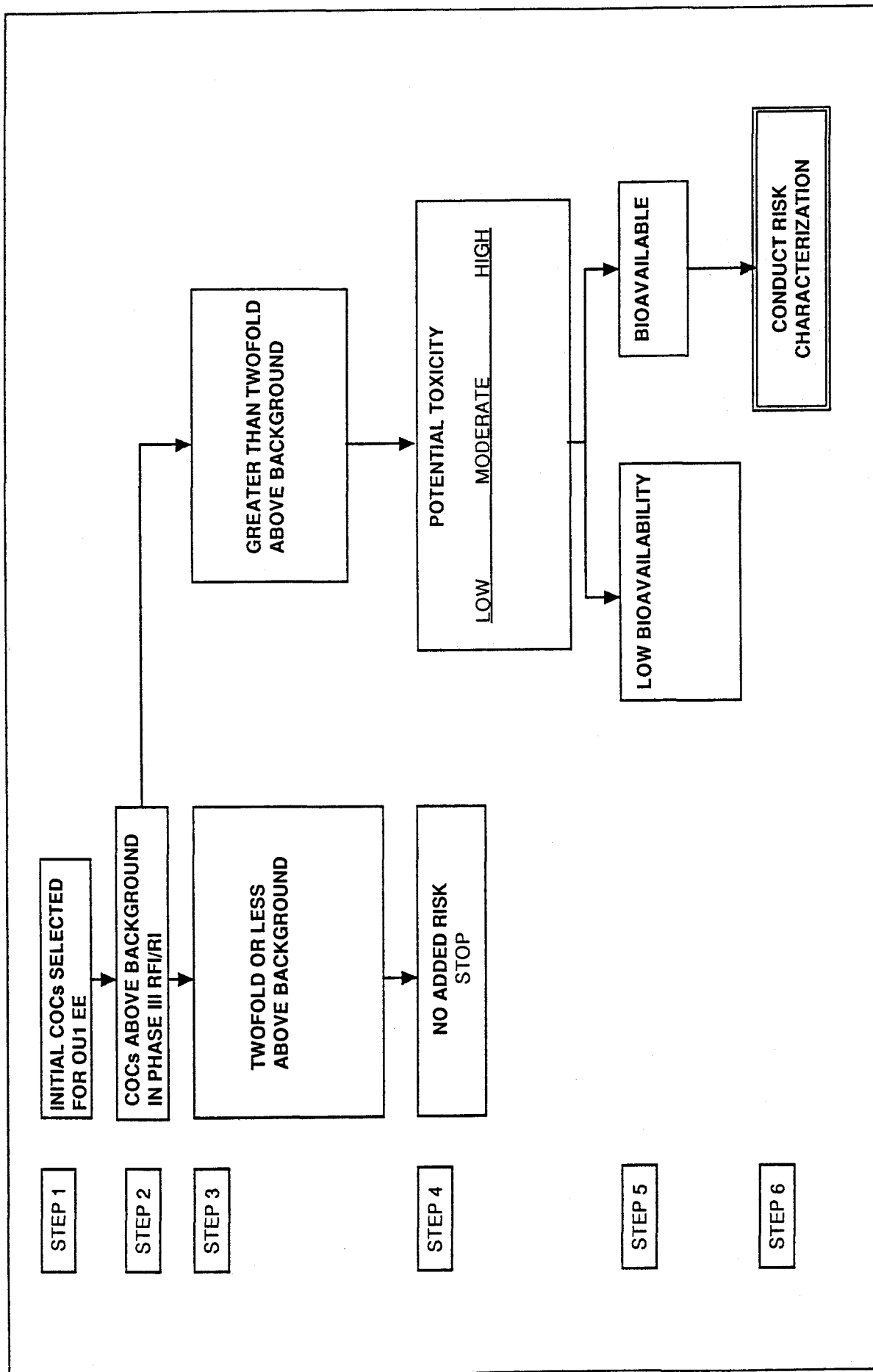
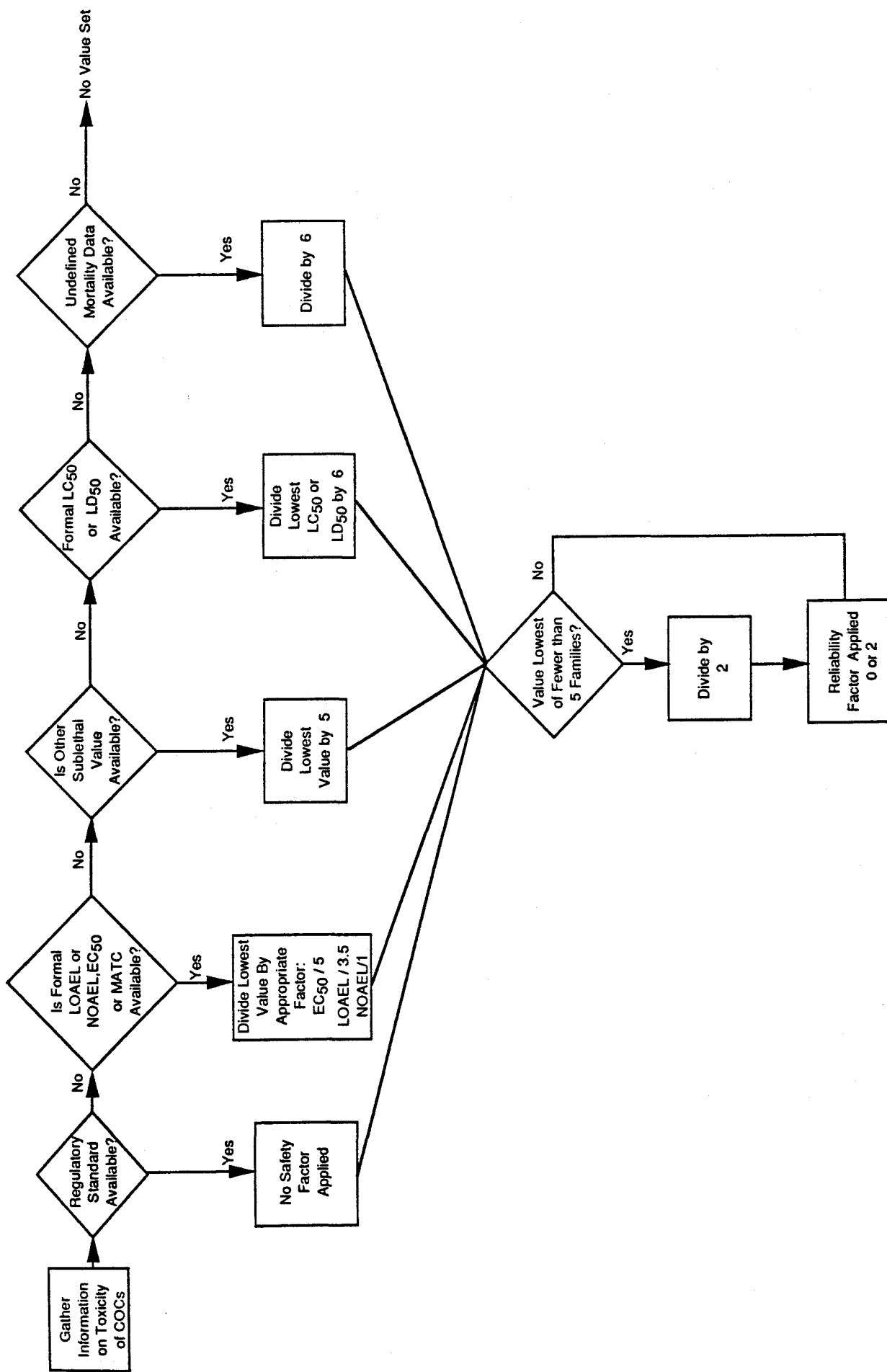


Figure E 3.5-1. Screening Process to Identify Contaminants with Potential Ecological Impacts at OU1 within Rocky Flats Plant



TRV Set

## E 3.5-2. Decision Process for Determination of Toxicity Reference Values (TRVs)

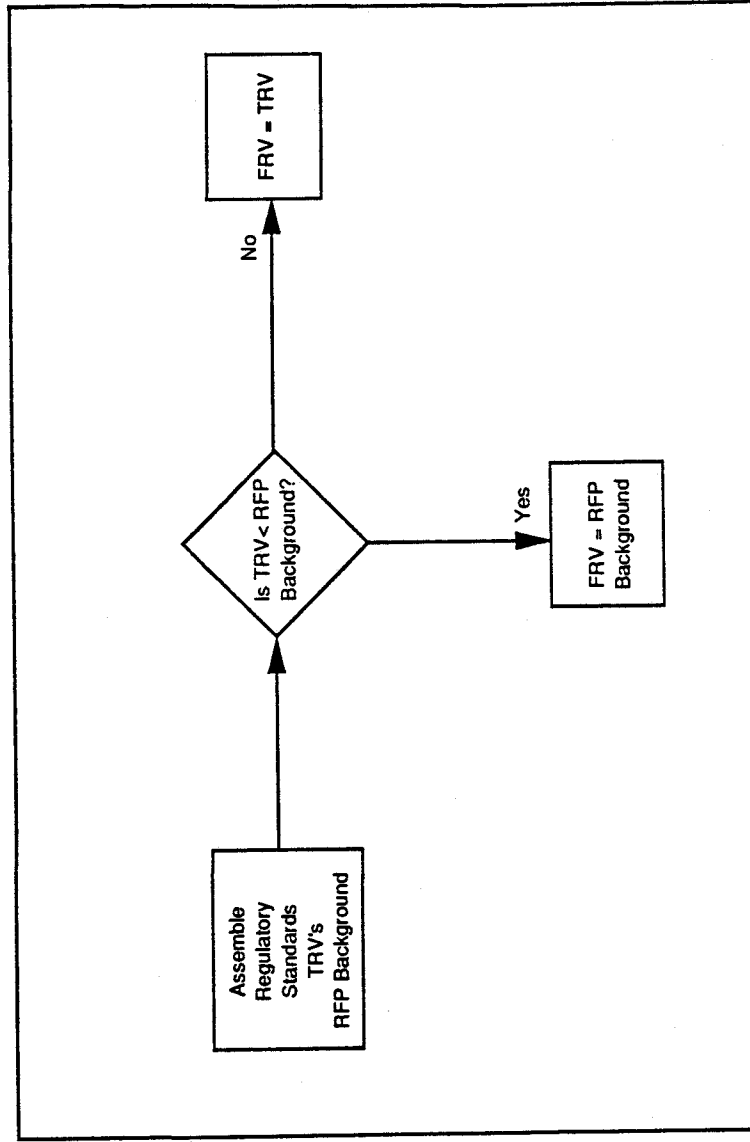


Figure E 3.5-3. Decision Process for Determination of Final Reference Values (FRVs)

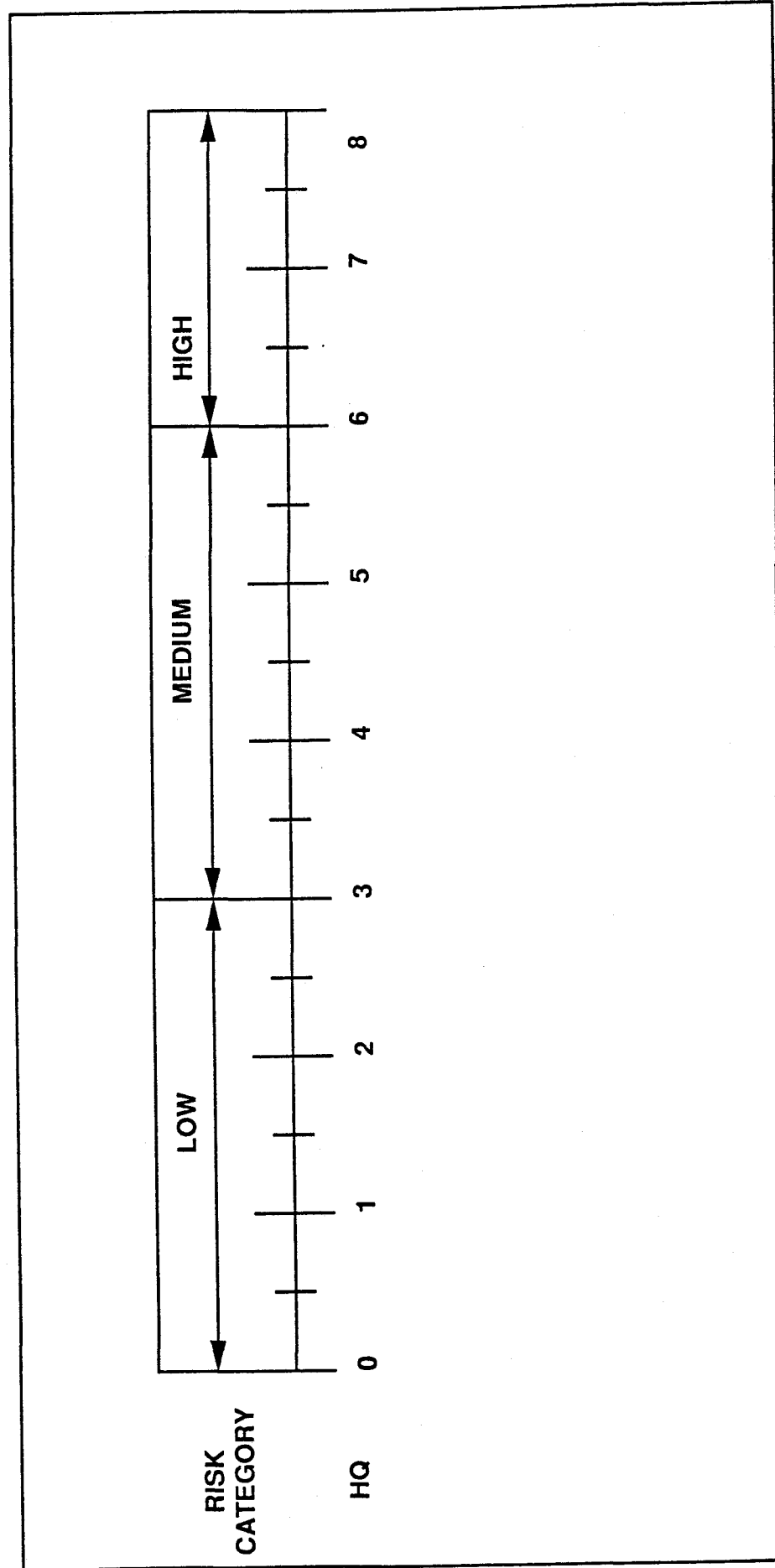


Figure E 3.7-1 Relationship Between Environmental Risk Level and Hazard Quotient (HQ)

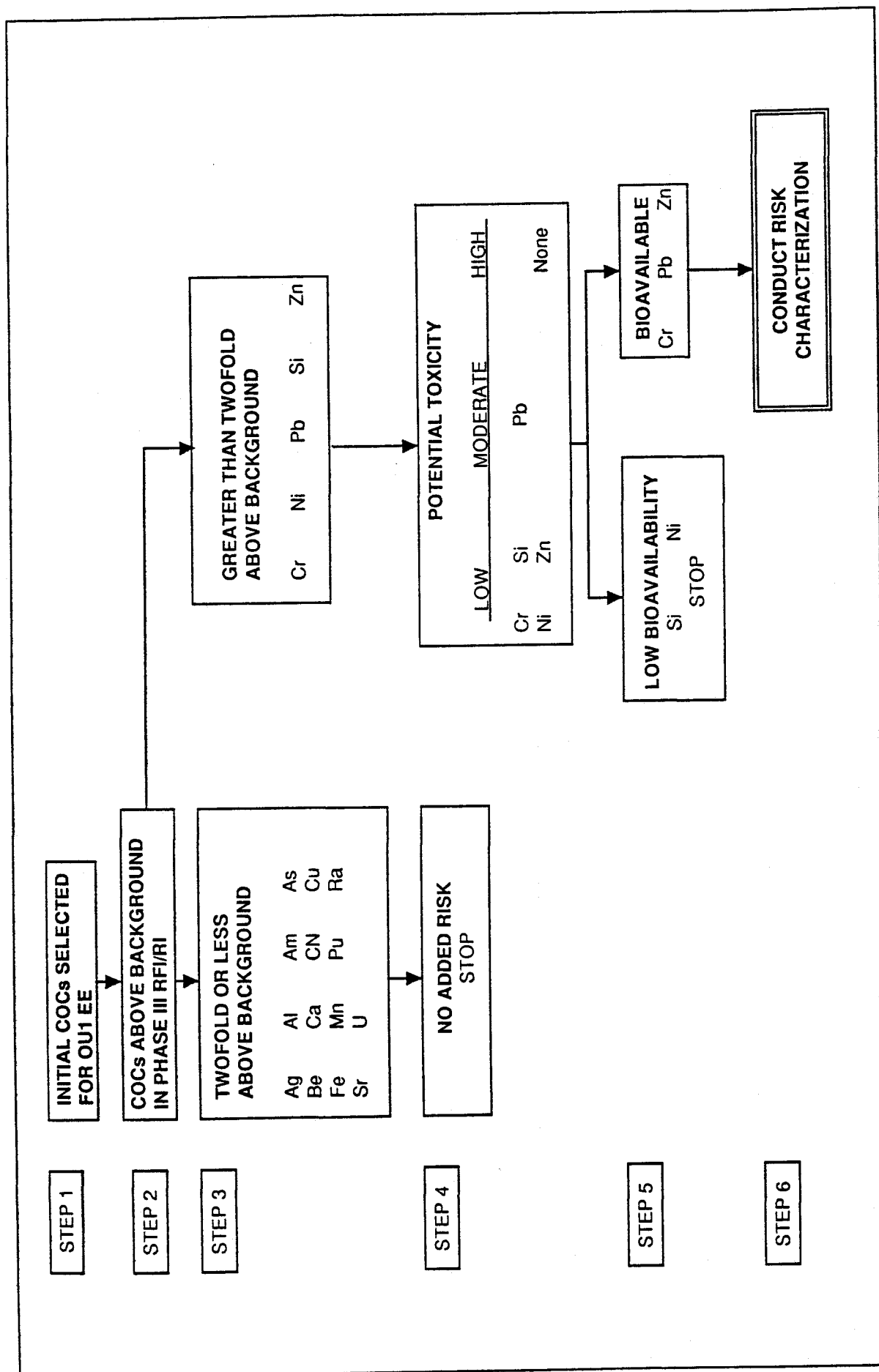


Figure E 4.1-1. Screening Process for Soils to Identify Contaminants with Potential Ecological Impacts at OU1 within Rocky Flats Plant

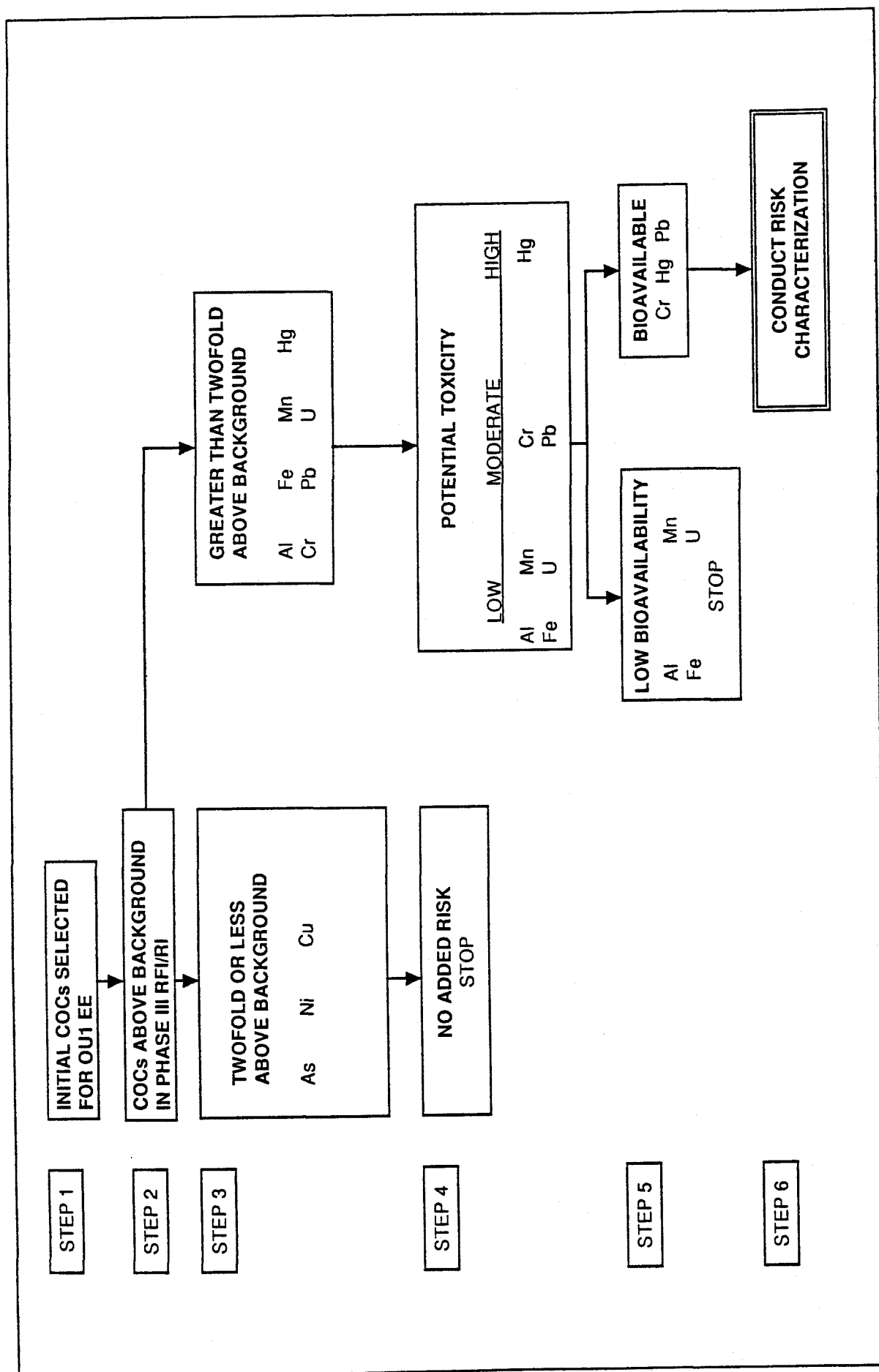


Figure E 4.1-2. Screening Process for Surface Waters to Identify Contaminants with Potential Ecological Impacts at OU1 within Rocky Flats Plant

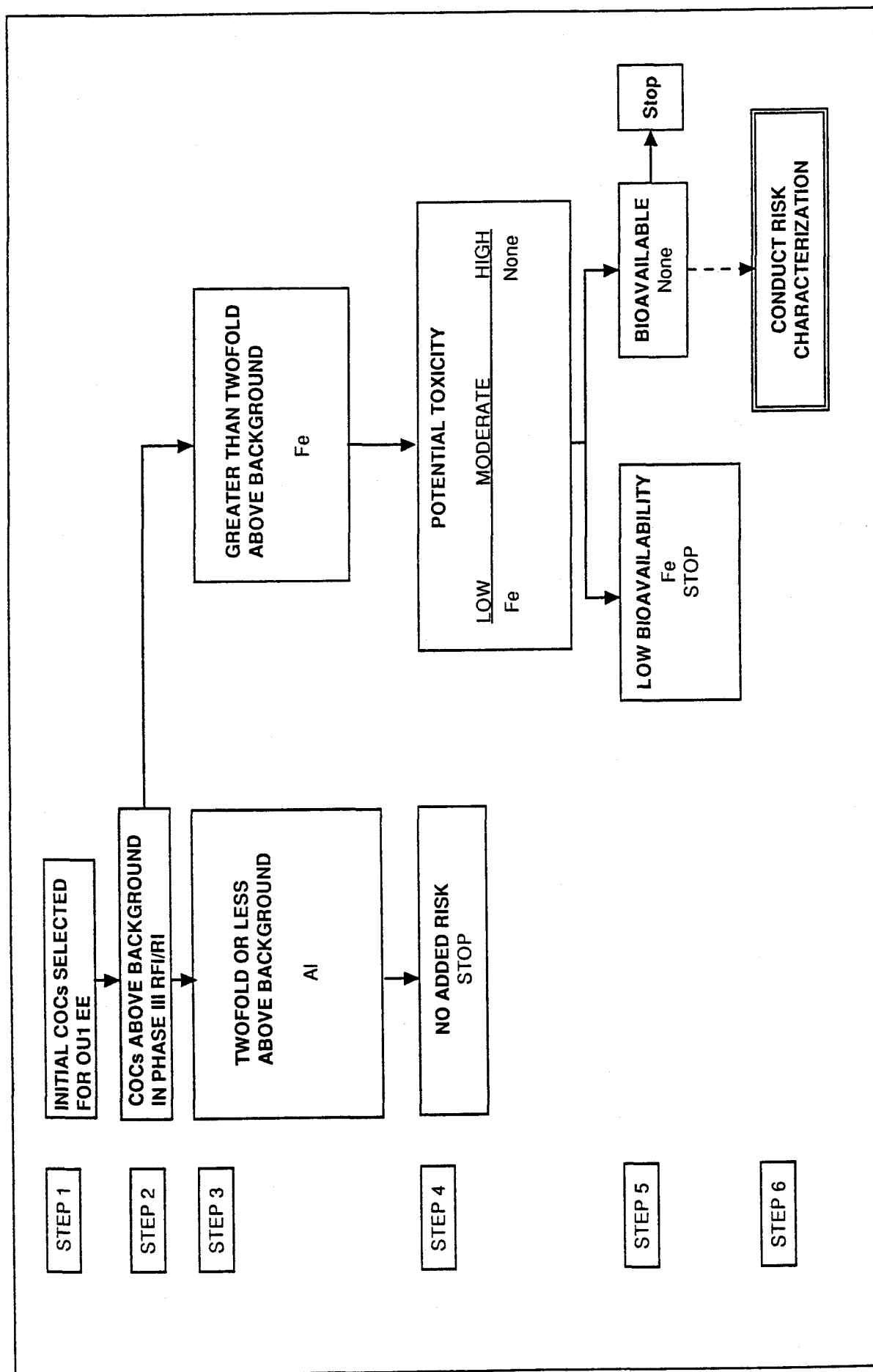


Figure E 4.1-3. Screening Process for Sediments to Identify Contaminants with Potential Ecological Impacts at OU1 within Rocky Flats Plant



**Appendix E**  
**Attachments**

**Environmental**  
**Evaluation**

**Phase III**  
**RF/RI Report**

U1 OU1 OU  
881 Hillside  
OU1 OU1 O  
Hillside 881  
1 OU1 OU1  
side 881 Hill  
U1 OU1 OU  
881 Hillside  
OU1 OU1 O  
Hillside 881

**ATTACHMENT E.A**

**POTENTIAL TOXICITY OF CONTAMINANTS OF CONCERN**

## TABLE OF CONTENTS

<u>Section</u>	<u>Page</u>
<b>1.0 INTRODUCTION</b> .....	EA-1
<b>2.0 ALUMINUM</b> .....	EA-2
2.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-2
2.2 TERRESTRIAL INVERTEBRATES .....	EA-3
2.3 TERRESTRIAL VERTEBRATES .....	EA-4
2.4 AQUATIC FAUNA .....	EA-4
2.5 REFERENCES CITED FOR ALUMINUM .....	EA-5
<b>3.0 ARSENIC</b> .....	EA-6
3.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-7
3.2 TERRESTRIAL INVERTEBRATES .....	EA-9
3.3 TERRESTRIAL VERTEBRATES .....	EA-9
3.4 AQUATIC FAUNA .....	EA-10
3.5 REFERENCES CITED FOR ARSENIC .....	EA-12
<b>4.0 BERYLLIUM</b> .....	EA-14
4.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-15
4.2 TERRESTRIAL INVERTEBRATES .....	EA-16
4.3 TERRESTRIAL VERTEBRATES .....	EA-16
4.4 AQUATIC FAUNA .....	EA-16
4.5 REFERENCES CITED FOR BERYLLIUM .....	EA-17
<b>5.0 CADMIUM</b> .....	EA-18
5.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-18
5.2 TERRESTRIAL INVERTEBRATES .....	EA-19
5.3 TERRESTRIAL VERTEBRATES .....	EA-25
5.4 AQUATIC FAUNA .....	EA-28
5.5 REFERENCES CITED FOR CADMIUM .....	EA-30
<b>6.0 CHROMIUM</b> .....	EA-34
6.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-34
6.2 TERRESTRIAL INVERTEBRATES .....	EA-35
6.3 TERRESTRIAL VERTEBRATES .....	EA-36
6.4 AQUATIC FAUNA .....	EA-37
6.5 REFERENCES CITED FOR CHROMIUM .....	EA-37

TABLE OF CONTENTS  
(continued)

<u>Section</u>	<u>Page</u>
<b>7.0 COPPER</b> .....	EA-40
7.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-40
7.2 TERRESTRIAL INVERTEBRATES .....	EA-41
7.3 TERRESTRIAL VERTEBRATES .....	EA-44
7.4 AQUATIC FAUNA .....	EA-45
7.5 REFERENCES CITED FOR COPPER .....	EA-46
<b>8.0 CYANIDE</b> .....	EA-49
8.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-49
8.2 TERRESTRIAL INVERTEBRATES .....	EA-50
8.3 TERRESTRIAL VERTEBRATES .....	EA-50
8.4 AQUATIC FAUNA .....	EA-50
8.5 REFERENCES CITED FOR CYANIDE .....	EA-51
<b>9.0 IRON</b> .....	EA-51
9.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-52
9.2 TERRESTRIAL INVERTEBRATES .....	EA-53
9.3 TERRESTRIAL VERTEBRATES .....	EA-54
9.4 AQUATIC FAUNA .....	EA-54
9.5 REFERENCES CITED FOR IRON .....	EA-55
<b>10.0 LEAD</b> .....	EA-56
10.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-57
10.2 TERRESTRIAL INVERTEBRATES .....	EA-58
10.3 TERRESTRIAL VERTEBRATES .....	EA-63
10.4 AQUATIC FAUNA .....	EA-65
10.5 REFERENCES CITED FOR LEAD .....	EA-66
<b>11.0 MANGANESE</b> .....	EA-69
11.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-70
11.2 TERRESTRIAL INVERTEBRATES .....	EA-71
11.3 TERRESTRIAL VERTEBRATES .....	EA-71
11.4 AQUATIC FAUNA .....	EA-71
11.5 REFERENCES CITED FOR MANGANESE .....	EA-72

TABLE OF CONTENTS  
(continued)

<u>Section</u>	<u>Page</u>
<b>12.0 MERCURY</b> .....	EA-74
12.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-74
12.2 TERRESTRIAL INVERTEBRATES .....	EA-75
12.3 TERRESTRIAL VERTEBRATES .....	EA-75
12.4 AQUATIC FAUNA .....	EA-76
12.5 REFERENCES CITED FOR MERCURY .....	EA-77
<b>13.0 SILVER</b> .....	EA-78
13.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-79
13.2 TERRESTRIAL INVERTEBRATES .....	EA-80
13.3 TERRESTRIAL VERTEBRATES .....	EA-80
13.4 AQUATIC FAUNA .....	EA-80
13.5 REFERENCES CITED FOR SILVER .....	EA-81
<b>14.0 ZINC</b> .....	EA-82
14.1 TERRESTRIAL AND AQUATIC VEGETATION .....	EA-82
14.2 TERRESTRIAL INVERTEBRATES .....	EA-83
14.3 TERRESTRIAL VERTEBRATES .....	EA-87
14.4 AQUATIC FAUNA .....	EA-88
14.5 REFERENCES CITED FOR ZINC .....	EA-89

ATTACHMENT E.A  
POTENTIAL TOXICITY OF CONTAMINANTS OF CONCERN

## 1.0 INTRODUCTION

The purpose of Attachment E.A. is to summarize the literature on toxicity and potential for biomagnification of the 20 contaminants of concern with regard to four groups of organisms: terrestrial and aquatic vegetation, terrestrial invertebrates, terrestrial vertebrates, and aquatic fauna. These data were used to develop reference values for Section E3.5. The information was also used to evaluate exposure and potential hazards as a result of the exposures at Rocky Flats Plant (RFP).

The scientific literature includes numerous examples of wild populations of organisms that have been exposed to potentially toxic trace elements from the deposition of atmospheric pollutants or from disposal on land of mining and other industrial wastes. Assessing the impacts of these contaminants is complex for a number of reasons. The existing body of literature is variable, being extensive for some contaminants, such as cadmium, and sparse for others, such as cyanide and beryllium. In addition, the effects of contaminants on biota may vary with many factors, including pH, temperature, species, age, presence of other contaminants, season, food source, and acclimatization.

Acclimatization is the process by which organisms adjust to changes in response to environmental conditions. Over time, evolutionary processes may produce select adaptations to improve a species tolerances to contaminants. The mechanisms for acquiring resistance include formation of metallothioneins, low molecular weight proteins that bind certain heavy metals such as cadmium, and other internal sequestering mechanisms. Behavioral mechanisms, such as avoidance, may also play a role. High background levels of many of the contaminants of concern may have resulted in acclimatization. As a result, whole body concentrations in these wild populations may be higher than levels that would be toxic to either laboratory organisms or other wild populations not exposed to such high background levels.

Development of the U.S. Environmental Protection Agency (EPA) Ambient Water Quality Criteria was based on "acid-soluble" fractions of metals in water. EPA has not officially approved methods for acid extraction of metals and recommends that "total recoverable" metals be used in evaluating attainment of water quality criteria (EPA 1985). It is important to recognize that this approach probably overestimates the bioavailable metal fraction, and therefore toxicity, because particulate, inorganic, and organic complexes, which are not available, are included in total recoverable fraction.

## 2.0 ALUMINUM

Aluminum (Al) is abundant in the earth's crust and is ubiquitous in air, water, and soil. Toxic effects of Al on soil communities and higher plants have been documented. Little work has been done to determine its effects on terrestrial vertebrates and invertebrates despite concern that the toxic effects of Al may magnify through ecosystems.

### 2.1 TERRESTRIAL AND AQUATIC VEGETATION

Increased Al concentrations have been shown to be toxic to plants (Bartlett and Riego 1972, Runge 1984, Horst 1985). Aluminum is mobilized in terrestrial and aquatic ecosystems by increasing the acidity of soil and water. If the soil is not too strongly acidic, one or more of the water molecules ionize, releasing hydrogen,  $H^+$ , to the solution, thereby increasing its acidity. Generally, the media of concern for Al uptake in plants are soil and nutrient solutions.

The general effects of Al toxicity in plants are growth inhibition, including both shoots and roots. Taylor et al. (1991) subjected wheat (*Triticum aestivum*) to various Al levels in soil and found that 18 micromoles ( $\mu M$ ) was the toxicity threshold, with a 1.3 percent growth reduction per  $\mu M$ . The toxicity threshold is defined as the lowest concentration of the metal at which an additional dose will produce a reduction in yield.

Campbell et al. (1990) reported that toxic levels of Al can cause severe yield reduction in clover (*Trifolium pratense*), especially in the presence of drought stress. Comparisons were made

between tests using soil media (26.2 percent Al saturation [pH 4.8] versus 2.8 percent Al saturation [pH 5.7]) and nutrient solutions (0 versus 111  $\mu$ M Al [pH 4.5]). Clover yields were significantly reduced in the soil media but not in the nutrient solutions. Hydroponic solution experiments generally reveal higher thresholds for toxicity levels than soil media experiments.

Toxic exposure limits were determined for two algae, *Scenedesmus* sp. and *Chlorella* sp., using hydroponic solutions (Lindemann et al. 1990). The toxic exposure limit was 4  $\mu$ M, growth inhibition occurred at this concentration.

Alva and Summer (1989) determined that the toxicity of Al was substantially alleviated by the addition of phosphogypsum (PG) or  $\text{CaSO}_4 \cdot 2 \text{H}_2\text{O}$  to the soil. Toxic effects to soybeans (*Glycine max*), decreased with an increase in soil pH. Aluminum was extremely toxic to soybeans at 40  $\mu$ M. Plants had poor root growth in nutrient solutions with Al concentrations exceeding this concentration. *Macadamia* sp. showed toxic effects if leaf Al concentrations exceeded 275 milligrams per kilogram (mg/kg) (Nagao and Hirae 1992). This study found that Al availability and uptake is increased at lower soil pH, especially in halloysitic soils.

Aluminum exhibits great potential for toxic effects on plants if growth media are acidic. Aluminum can also be toxic at basic pH levels (Hesse 1971). Near pH 8, and reaching a maximum concentration near pH 10, two new soluble Al species,  $(\text{Al}[\text{H}_2\text{O}]_2[\text{OH}]_4)^{-1}$  and  $(\text{Al}[\text{H}_2\text{O}][\text{OH}]_5)^{-2}$  occur. However, most research has focused on acidic soils. Unfortunately, the majority of studies involving Al have been conducted on deficiency levels for agronomic plants, rather than on toxicity. The bioaccumulation of Al in plants and its movement through the food chain are not well known.

## 2.2 TERRESTRIAL INVERTEBRATES

As mentioned earlier, few studies have been conducted on the effects of Al on terrestrial invertebrates. One study, which examined the accumulation of heavy metals in earthworms in sewage sludge, found a significant negative regression of Al concentration on body weight:



higher Al concentration was associated with lower body weight. The causal mechanisms are not known.

### 2.3 TERRESTRIAL VERTEBRATES

Aluminum occurs naturally in vertebrates. However, very little work has been conducted on the effects of Al toxicity in terrestrial vertebrates and associated ecosystems. No evidence was found for biomagnification in terrestrial vertebrates. Brain tissue of mammals normally contains 1 to 2 micrograms per gram ( $\mu\text{g/g}$ ) Al (dry weight). The toxic range is reportedly 4 to 8  $\mu\text{g/g}$  in brain tissue for the cat and rabbit (Crapper et al. 1976).

Human studies have shown that Al compounds can affect absorption of other elements in the gastrointestinal tract and alter intestinal function. The binding of phosphorus in the gut can lead to phosphate depletion. Aluminum may alter gastrointestinal tract motility (Goyer 1986).

### 2.4 AQUATIC FAUNA

The toxicity of Al to aquatic fauna results primarily from the soluble inorganic forms present in ambient water. In aqueous solution, Al is amphoteric, with minimum solubility at about pH 5.5. At more acidic pH, Al toxicity is thought to be due to soluble monomeric forms ( $\text{Al}^{+3}$ ), and to either precipitates formed on respiratory surfaces or to aluminate ( $\text{Al}[\text{OH}]_4^{+2}$ ), the major ionic species above pH 6.5 (Leivestad et al. 1987, Tietge et al. 1988, Ingersoll et al. 1990a,b, Weatherly et al. 1991).

Acute toxicity of Al to fish, as judged by the 96-hour  $\text{LC}_{50}$ , ranges from 3,600 micrograms per liter ( $\mu\text{g/l}$ ) (total Al) for the brook trout (*Salvelinus fontinalis*) to greater than 35,000  $\mu\text{g/l}$  for several warmwater and coldwater species (EPA 1988). More incipient effects, such as behavioral modifications, reduced growth rate, and developmental impairment, have been observed at concentrations as low as 150  $\mu\text{g/l}$  at circumneutral pH in laboratory experiments. Many taxa, especially vertebrates, may detoxify heavy metals through binding to proteins, called metallothioneins, produced in response to heavy metal contamination. Therefore, natural

populations acclimatized to ambient Al concentrations may not be subject to toxic effects seen in laboratory animals.

Aquatic invertebrates, including mollusks, insects, crustaceans, and flatworms, have also been tested for their sensitivity to Al. The range of concentrations toxic to invertebrates overlaps that of fish (EPA 1988). The LC<sub>50</sub> for Al chloride or Al sulfate exceeds 22,000 µg/l for most species tested, including the planarian *Dugesia tigrina* (Platyhelminthes), the snail genus *Physa*, the amphipod *Gammarus pseudolimnaeus*, the stonefly genus *Acroneuria* (Plecoptera), and the midge *Tanytarsus dissimilis* (Chironomidae). The most sensitive invertebrate tested was the clodoceran *Ceriodaphnia dubia*, for which the mean acute value was 2,648 µg/l. Sublethal effects such as immobilization and reproductive impairment were detected at concentrations as low as a few hundred µg/l.

Geological formations at RFP have a high clay content, which leads to high natural concentrations in soils, sediment, and surface water. The background Al concentrations of Al in surface water at RFP are reported as 485 µg/l (dissolved Al) and 60.42 µg/l (total recoverable Al). Colorado State stream standards are 950 µg/l for acute exposures and 150 µg/l for chronic exposures. The background concentrations for surface water at RFP exceed the state's chronic standard. However, the lower standard was suggested to protect sensitive species such as brook trout and striped bass (EPA 1988), which do not occur at RFP. The toxicity reference value (TRV) for Al set at the RFP background values of 485 µg/l (dissolved) and 60.42 µg/l (total) should be protective of most fish and aquatic invertebrates at the site.

## 2.5 REFERENCES CITED FOR ALUMINUM

- Alva, A.K. and M.E. Sumner. 1989. "Alleviation of aluminum toxicity to soybeans by phosphogypsum or calcium sulfate in dilute nutrient solutions." Soil Science 147:278-285.
- Bartlett, R.J. and D.C. Riego. 1972. "Toxicity of hydroxy aluminum in relation to pH and phosphorus." Soil Science 114:194-200.
- Campbell, T.A., N.J. Nuernberg, and C.D. Foy. 1990. "Differential responses of red clover germplasms to aluminum stress." Journal of Plant Nutrition 13(11):1463-1474.

Crapper, D.R., S.S. Krishnan, and S. Quittkat. 1976. "Aluminum, neurofibrillary degeneration and Alzheimer's disease." Brain 99:69-79.

EPA. 1988.

Goyer, R.A. 1986. "Toxic effects of metals." In Toxicology: the basic science of poisons (C. D. Klaassen, M. O. Amdur, and J. Doull, eds.). Macmillan Publishing Company, New York.

Hesse, P.R. 1971. A Textbook of Soil Chemical Analysis. Chemical Publishing Inc., New York.

Horst, W.J. 1985. "Effect of aluminum on root growth, cell division rate and mineral element contents in roots of *Vigna unguiculata* genotypes." Z. Pflanzenernahr. Bodenk 148:335-348.

Ingersoll et al. 1990a.

Ingersoll et al. 1990b.

Leivestad et al. 1987.

Lindemann, J., E. Holtkamp, and R. Herrmann. 1990. "The impact of aluminum on green algae isolated from two hydrochemically different headwater streams, Bavaria, Germany." Environmental Pollution 67(1):61-78.

Nagao, M.A. and H.H. Hirae. 1992. "Macadamia: cultivation and physiology." Critical Reviews in Plant Sciences 10(5):441-470.

Runge, M. 1984. "Bedeutung und Wirkung von Aluminum als Standortfaktor." Dusseldorfer Geobotanische Kolloquien 1:3-10.

Taylor, G.J., K.J. Stadt, and M.R.T. Dale. 1991. "Modelling the phytotoxicity of aluminum, cadmium, copper, manganese, nickel, and zinc using the Weibull frequency distribution." Canadian Journal of Botany 69(2):359-367.

Tietge et al. 1988.

Weatherly et al. 1991.

### 3.0 ARSENIC

Arsenic (As) occurs naturally in living organisms, but no confirmed physiological function has been attributed to this metal. Arsenic exhibits two inorganic forms with differing toxic

properties. Arsenic III is known to bind to sulfhydryl groups on proteins, disrupting their function. The mechanism of As V toxicity is less well known, but it does not bind to sulfhydryl groups. It does appear to selectively uncouple oxidative phosphorylation, poisoning aerobic ATP generation. Under acidic conditions, As III will slowly oxidize to As V.

### 3.1 TERRESTRIAL AND AQUATIC VEGETATION

Arsenic occurs in virtually all soils and natural waters. Plants have therefore evolved in the presence of As ions, and it is possible that As is an essential element for plant growth. However, beneficial effects of As on plants have been documented. Arsenic is chemically similar to phosphorus, which is an essential plant nutrient.

Arsenic accumulation in plants is variable, depending on solubility of the arsenicals and soil properties. If sufficient As is absorbed, plants may be killed. Alternatively, As may accumulate in plant biomass and enter the food chain (Treshow 1978).

The uptake mechanism of As to plants has been reported (NAS 1977). When As in solution penetrates the cuticle of the root and enters the apoplast system, it bathes the external surface of plasmalemma of the symplast. This is the location of at least some of the enzymes of the living plant. One of the first symptoms of injury by sodium arsenite is wilting (loss of turgor), which suggests an alteration in membrane integrity or permeability. Arsenites are more toxic than arsenates. The arsenate symptoms include yellowing (chlorosis), but not rapid loss of turgor. Arsenate is known to uncouple phosphorylation, thus impeding the availability of ATP to the plant. Arsenic and its derivatives are most commonly used in plant herbicides because of these pathway effects.

Callahan and Shepard (1991) studied the toxic effects of As on large crabgrass (*Digitaria sanguinalis*), annual bluegrass (*Poa annua*), and creeping bentgrass (*Agrostis palustris*) via the soil. Poor germination of seeds and poor regrowth of adult plants was observed for annual bluegrass and large crabgrass when concentrations of As totaled more than 136 kilograms per

hectare (kg/ha) (based on soil residue build-up). The same effects were observed for creeping bentgrass when concentrations in the soil were greater than 272 kg/ha.

Generally, As is introduced into the environment and growth media as an organic arsenical (i.e., herbicide). The NAS (1977) found that As at soil concentrations of 1-4 parts per million (ppm) caused shotholing and defoliation of leaves in peach trees. Woolson (1973) studied uptake and phytotoxicity in green beans, lima beans, spinach, cabbage, tomatoes, and radishes. These vegetables had no growth in soil with As concentrations of 500 ppm, where As was applied via spray solutions to the plants and the soil.

Most studies have focused on the relationship between As and phosphorus. Everett (1962) indicated that phosphorus increased the As content of bluegrass and crabgrass in a turf treated with tricalcium arsenate. However, he found that phosphorus reduced absorption of tricalcium arsenate (measured as As) from nutrient solutions from 246 to 29 ppm. He found a potential species difference from the results of his study. Sckerl (1968) found that phosphorus reduced As toxicity.

Arsenic also seems to have an interactive relationship with zinc. Batjer and Benson (1958) showed that toxicity in peaches grown in As-contaminated soils could be reduced by foliar applications of zinc or iron chelates or soil applications of zinc or iron sulfates. The relationship is not completely clear, but Burleson et al. (1967) suggested that, with absorption of more than optimal phosphorus, phosphorus, and zinc reacted together in a manner that reduced either their mobility or their solubility. There may be an interactive relationship between As, phosphorus, and zinc that enhances or minimizes toxicity of As in soil to plants.

Very little food web modeling has been performed with As as a primary analyte. Arsenic seems to be of more interest for its interactive properties with other plant nutrients. The distribution of As through the food chain is greatly limited by its phytotoxic effects. That is, plant injury would generally occur if concentrations toxic to wildlife could be reached.

### 3.2 TERRESTRIAL INVERTEBRATES

Historically, As has been used as an insecticide and molluscicide. Its toxicity to soil fauna has been observed in agricultural areas treated with arsenical pesticides. The wide spectrum toxicity of As stems from its ability to block the TCA or citric acid cycle, a basic metabolic pathway of all higher organisms. Thus, it is potentially toxic to virtually all terrestrial invertebrates. Arsenic may also inhibit DNA repair systems; at lower concentrations, it may be toxic due to a mutagenic effect (Jernelov et al. 1978).

Given the widespread knowledge of the toxic effects of As on both plants and animals, it is surprising that so few studies have examined its effects on terrestrial food chains. Andren et al. (1973) found that As bioaccumulated in various levels of a deciduous forest ecosystem in eastern Tennessee. They reported soil and litter levels to be 2 and 0.8 ppm, respectively, rising to 1 and 11 ppm in leaves and roots, but falling to 0.3 and 0.1 ppm in branches and acorns. Composite samples of primary consumers showed some potential for accumulation: canopy feeding insects had As concentrations of 10 ppm. Both cryptozoans (litter dwelling) and earthworms (soil dwelling) showed higher As concentrations: 100 and 19 ppm, respectively. Unfortunately, the authors did not address feeding relationships in the ecosystem as a whole; thus, it is difficult to draw conclusions regarding the movement of As through a food web.

### 3.3 TERRESTRIAL VERTEBRATES

Little information is available on the effects, toxicity, and potential for accumulation as in terrestrial vertebrates. Absorption from the gastrointestinal tract is almost complete, as only 6 to 9 percent of orally administered As-labeled trivalent or pentavalent As is eliminated in feces in mice (Vahter and Norin 1980). Normal values for As in whole blood and urine in humans are less than 10 µg/l and 50 µg/l, respectively. Excessive exposure is 50 µg/l in whole blood and greater than 100 µg/l in urine (Goyer 1986).

A modest accumulation of As is seen in small mammals from orchards where lead As was used as a fungicide. White et al. (1977) reported concentrations in European starling (*Sturnus*

*vulgaris*) whole bodies (less skin, wings, and bill) of 0.019, 0.156, 0.171, and 0.139 µg/g. Details of experimental design were not provided with these data.

### 3.4 AQUATIC FAUNA

Little is known about the mechanisms of As toxicity to aquatic organisms; however, As readily forms stable bonds with sulfur and carbon in organic compounds. Enzyme inhibition is probably the primary mode of toxicity when As (III) reacts with sulfhydryl groups of proteins, and As (V) may uncouple oxidative phosphorylation (Fowler et al. 1977, Schiller et al. 1977). The chemistry of As in water is complex. Chemical, biochemical, and geochemical reactions control the concentration, oxidation state, and form of As in water (Braman 1983, Callahan et al. 1979, Holm et al. 1979, and Scudlark and Johnson 1982). Unlike many other heavy metals, the toxicity of As III in aquatic animals appears to be independent of water hardness (CaCO<sub>3</sub> content).

The four relevant As species common in natural waters are inorganic As (III) and As (V), methanearsonic acid, and dimethylarsinic acid. Previously, toxicity of As was evaluated in terms of total recoverable inorganic As (III). However, the more recent suggestion is that acid-soluble As (III) and acid-soluble As (V) are probably the best measurements for expressing aquatic life criteria for As (EPA 1985). Use of these acid-soluble measurements has both toxicological and practical advantages. The following data are expressed in these terms, unless otherwise noted.

At circumneutral pH, acute toxicity of As III to common freshwater fish has been reported at levels from 13,340 µg/l for rainbow trout (*Salmo gairdneri*) to over 41,000 µg/l for bluegill (*Lepomis macrochirus*) (EPA 1984). Sublethal effects of As III include reduced growth rate, behavioral alterations, decreased enzyme activity, alterations in blood chemistry, and decreased hematocrit (EPA 1984). Tissue damage and decreased survival were observed for the bluegill at about 700 µg/l (Gilderhus 1966, cited in EPA 1984). A freshwater final acute value (as defined by Stephan et al. 1985) of 718.2 µg/l for inorganic As (III) was determined from a literature review by EPA (1985).

Among the invertebrates, amphipods are the most sensitive, with a mean acute value of 874 µg/l for the genus *Gammarus*. Cladocerans are next in sensitivity, with mean acute values of about 1,000 to 2,500 µg/l for the genera *Simocephalus*, *Daphnia*, and *Ceriodaphnia*. Other invertebrate taxa are less sensitive. Acute toxicity values are 22,000 µg/l for the stonefly *Pteronarcys dorsata*; 24,500 for the snail *Aplexa hypnorum*; and 97,000 for the midge *Tanytarsus dissimilis* (EPA 1984).

In fish, acute toxicity to monosodium methanearsonate ranges from an LC<sub>50</sub> of 1,921 µg/l for the bluegill to an LC<sub>50</sub> of 1,403,000 µg/l for the channel catfish (*Ictalurus punctatus*) (Anderson et al. 1975, Johnson and Finley 1980).

Chronic toxicity of inorganic As (III) was evaluated with *Daphnia magna* (Call et al. 1983, Lima et al. 1984). In a life-cycle test, chronic values of 914.1 µg/l were found, based on chronic limits of 633.0 and 1,320 µg/l. The 96-hr LC<sub>50</sub> for this species in the same study was 4,340 µg/l.

Chronic toxicity to As (V) was tested on early life-stages of the fathead minnow (*Pimephales promelas*) (DeFoe 1982); the chronic value was 891.6 µg/l. The 96-hr LC<sub>50</sub> for this species was 25,600 µg/l. Although not measured in terms of the suggested acid-soluble As mentioned earlier, an acute value of 7,400 µg/l was found in a life-cycle test with *Daphnia magna* (Biesinger and Christensen 1972).

Tests with early life stages appeared to be the most sensitive indicator of As toxicity. The lowest value obtained in any test on As was 40 µg/l from a 7-day exposure of embryos and larvae of the toad *Gastrophryne carolinensis* to inorganic As (III) (Birge 1978). This value is about a factor of 4.5 lower than the freshwater final chronic value (as defined by Stephan et al. 1985) for inorganic As (III).

Following the criteria and procedures of Stephan et al. (1985), freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of As (III)



does not exceed 190 µg/l more than once every three years on the average and if the one-hour average concentration does not exceed 360 µg/l more than once every three years on the average (EPA 1985). A locally important species with high sensitivity would be an exception.

Bioconcentration tests have been conducted on freshwater fish and invertebrates. The highest bioconcentration factor (BCF) was 17, which was obtained for inorganic As (III) with a snail (Spehar et al. 1980). An early life-stage test on As (V) with the fathead minnow (Defoe 1982) showed that the bioconcentration decreased with increased exposure. Bioconcentration factors were slightly lower (down to 1.2) in exposure concentrations that caused significant adverse effects than those that did not (EPA 1985). Pretreatment of rainbow trout to As (III) enhanced the elimination of a subsequent dose of As. Additional results indicated that fish retained less As after four weeks of exposure than after two weeks (Oladimeji et al. 1982). In a number of cases, As toxicity for fish increased with increased duration of exposure (EPA 1985).

The freshwater residue data indicate that As is not bioconcentrated to a high degree but that lower forms of aquatic life may accumulate higher As residues than fish. The low bioconcentration factor and short half-life of As in fish tissue suggest that residues should not be a problem to predators of aquatic life (EPA 1985).

### 3.5 REFERENCES CITED FOR ARSENIC

Anderson, A.C., et al. 1975. "The acute toxicity of MSMA to black bass (*Micropterus dolomieu*), crayfish (*Procambarus* sp.), and channel catfish (*Ictalurus lacustris*)." Bull. Environ. Contam. Toxicol. 14:330.

Andren, A.W., J.A. Fortescue, G.S. Henderson, D.E. Reichle, and R.I. Hook. 1973. In: Ecology and Analysis of Trace Contaminants - Progress Report June 1972 to January 1973. Pp. 61-119. ORNL-NSF-EATC-1, Oak Ridge National Laboratory, TN.

Batjer, L.P. and N.R. Benson. 1958. "Effect of metal chelates in overcoming arsenic toxicity to peach trees." Proc. Amer. Soc. Hort. Sci. 72:74-78.

Biesinger, K.E. and G.M. Christensen. 1972. "Effects of various metals on survival, growth, reproduction, and metabolism of *Daphnia magna*." Jour. Fish. Res. Board Can. 29:1691.

Birge, W.J. 1978. "Aquatic toxicology of trace elements of coal and fly ash." In: Energy and Environmental Stress in Aquatic Systems (J.H. Thorp and J.W. Gibbons, eds.). CONF-771114. National Technical Information Service. Springfield, Va. 219 pp.

Braman, R.S. 1983. "Environmental reaction and analysis methods." In: Biological and Environmental Effects of Arsenic (B.A. Fowler, ed.). Elsevier, New York. 141 pp.

Burleson, C.A. and N.R. Page. 1967. "Phosphorus and zinc interactions in flax." Soil Sci. Soc. Amer. Proc. 31:510-513.

Call, D.J., et al., 1983. Toxicity and metabolism studies with EPA priority pollutants and related chemicals in freshwater organisms. PB83-263665. National Technical Information Service. Springfield, Va.

Callahan, L.M. and D.P. Shepard. 1991. "Control of annual weedy grasses in a bentgrass green with treatment programs of tri-calcium arsenate." J. Am. Soc. Hortic. Sci. 116(1):30-35.

Callahan, M.A., et al., 1979. "Water-related environmental fate of 129 priority pollutants." Vol. I. EPA-440/4-79-029a. National Technical Information Service, Springfield, Va.

DeFoe, D.L. 1982. Memorandum to Robert L. Spehar. U.S. EPA, Duluth, Minn. July 9.

EPA. 1985. Ambient water quality criteria for arsenic - 1984. Office of Water Regulations and Standards, Criteria and Standards Division, EPA 440/5-84-033. 66 pp.

Everett, C.F. 1962. "Effect of phosphorus on the phytotoxicity of tricalcium arsenate as manifested by bluegrass and crabgrass." Ph.D. Thesis Rutgers University. New Brunswick, N.J.

Fowler, B.A., et al., 1977. "Ultrastructural and biochemical effects of prolonged oral arsenic exposure on liver mitochondria of rats." Environ. Health Perspect 19:197.

Goyer, R.A. 1986. "Toxic effects of metals." In Toxicology: the basic science of poisons (C. D. Klaassen, M. O. Amdur, and J. Doull, eds.). Macmillan Publishing Company, New York.

Holm, T.R., et al., 1979. "Heterogeneous interactions of arsenic in aquatic systems." In: Chemical Modeling in Aqueous Systems (E.A. Jenne, ed.). ACS Symposium Series No. 93. American Chemical Society. Washington, D.C. 712 pp.

Jernelov, A., K. Beijer, L. Soderlund. 1978. "General aspects of toxicology." In: Principles of Ecotoxicology (G.C. Butler, ed.). SCOPE 12. J. Wiley & Sons. New York.

Johnson, W.W. and M.T. Finley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. Resource Publication 137. U.S. Fish and Wildlife Service, Washington, D.C.

Lima, A.R., et al., 1984. "Acute and chronic toxicities of arsenic (III) to fathead minnows, flagfish, daphnids, and an amphipod." Arch. Environ. Contam. Toxicol. 13:595.

National Academy of Science. 1977. Arsenic. NAS. Washington, D.C.

Oladimeji, A.A., et al., 1982. "Effect of acclimation of rainbow trout (*Salmo gairdneri*) to arsenic on retention of a subsequent dose of arsenic." Ecotoxicol. Environ. Safety 6:196.

Schiller, C.M. et al., 1977. "Effects of arsenic on pyruvate dehydrogenase activation." Environ. Health Perspect 19:205.

Sckerl, M.M. 1968. "Translocation and metabolism of MAA-Carbon-14 in Johnsongrass and Cotton." Ph.D. Thesis. University of Arkansas. Fayetteville.

Scudlark, J.R. and D.L. Johnson. 1982. "Biological oxidation of arsenite in seawater." Estuarine Coastal Shelf Sci. 14:693.

Spehar, R.L., et al., 1980. "Comparative toxicity of arsenic compounds and their accumulation in invertebrates and fish." Arch. Environ. Contam. Toxicol. 9:55.

Stephan, C.E., et al., 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. National Technical Information Service. Springfield, Va.

Treshow, M. 1978. "Terrestrial plants and plant communities." In: Principles of Ecotoxicology (G.C. Butler, ed.). SCOPE 12. J. Wiley & Sons. New York.

Vahter and Norin. 1980.

White, D.H., J.R. Bean, and J.R. Longcore. 1977. "Nationwide residues of mercury, lead, cadmium, arsenic, and selenium in starlings, 1973." Pestic. Monitor J. 11:35-39.

Woolson. 1973.

#### 4.0 BERYLLIUM

Coal combustion is a major source of beryllium (Be) in the environment. Other sources are Be extraction plants, ceramic plants, Be alloy manufacturers, and manufactures of nuclear reactors, aircraft, and rockets. Of the contaminants of concern, Be, a light trace metal, and mercury, a heavy trace metal, have been defined as hazardous by EPA. This means that slight exposure can endanger human health.

#### 4.1 TERRESTRIAL AND AQUATIC VEGETATION

Ordinarily, the amount of Be found in vegetation is low. Soluble compounds of Be are taken up by the roots, especially if the soil is acidic. In many plants, the translocation of Be to shoots is poor. Among those plants that show poor translocation of Be are the bean, barley, sunflower, and tomato. Corn, however, translocates Be readily. A mechanism apparently exists by which plants eliminate Be. If Be is contained in fallen, dead leaves it is eliminated from the system (Wilber 1980).

Specific poisonous effects from Be are delayed germination, growth retardation, and growth suppression. Beryllium has been shown to increase the frequency of chromosome aberrations induced in barley by ethyl methanesulfonate (Oak Ridge National Laboratory 1978). Retardation of growth is the most typical response of plants to high levels of soil Be.

Beryllium is harmful to bush beans in concentrations of 1 ppm in soil media. As Be concentration in the soil increases, the rate of growth of the bush beans is more greatly suppressed (Durocher 1969). Many plants showed growth inhibition when exposed to approximately 2 ppm Be (Wilber 1980).

The EPA (1975) reported the presence of Be in raw waste effluents from the mining and milling of bertrandite. They found that Be in nutrient solutions, at acid pH is highly toxic to plants. Solutions containing 15 to 20 milligrams per liter (mg/l) of Be delay germination and retard growth of cress and mustard seeds in solution culture. Wilber (1980) also found that Be in experimental nutrient solutions in amounts greater than 1 to 2 ppm induced growth retardation in bush beans, tomatoes, alfalfa, barley, lettuce, and peas.

Beryllium is considered a nutrient-depleting agent (Gregory 1964) and has been shown to cause a decrease in copper content of the plant. Thus, some of the symptoms of Be toxicity may actually be associated with copper deficiencies in the plant. Very little research has been

conducted on Be toxicity to plants. What is known, however, is that plants bioaccumulate only trace amounts of Be (Wilber 1980).

#### 4.2 TERRESTRIAL INVERTEBRATES

No literature was located which addressed the effects of Be on terrestrial invertebrates.

#### 4.3 TERRESTRIAL VERTEBRATES

Beryllium is not considered essential to animals, and little is known about the effects of this trace element on wild animals. The primary vector for Be bioaccumulation in animals is inhalation. Tucker (1972) reports that there is little information about Be in natural food chains.

Major toxicologic effects on the lungs may result from inhalation of Be, as documented during studies in the laboratory rat. Clearance of inhaled Be is multiphasic; half is cleared in about two weeks. The remainder is removed slowly, and a residuum becomes fixed in tissues (Goyer 1986). Beryllium sulfate inhibits the absorption of glucose in the intestines of rats. Dental calcium is reduced in dogs fed a supplement of 1 to 3 grams of Be carbonate daily (Wilber 1980).

Beryllium is not excreted from mammalian tissue; consequently, its effects are cumulative. Biochemically, Be competes with magnesium for enzyme sites and has been shown to inhibit DNA polymerase, thymidine kinase, and alkaline phosphatase (Duffus 1980).

#### 4.4 AQUATIC FAUNA

As a general rule, little Be is present in aquatic media. What little is present apparently has minimal effect on aquatic organisms (Wilber 1980). Beryllium toxicity to aquatic organisms is influenced by several factors. Wilber (1971, cited in Wilber 1980) found that the toxicity to fish of a variety of wastewaters varies depending on the pH alkalinity, and hardness of the water.

The EPA (1978, cited in Wilber 1980) states that, in the freshwater environment, Be is acutely poisonous to fish at a concentration as low as 87 µg/l. Beryllium is said to be chronically poisonous to *Daphnia* at a concentration of 3 µg/l. The quality of the water significantly modifies the acute toxicity of Be. Generally, Be has a low solubility in water and is therefore somewhat unavailable to most aquatic organisms.

Beryllium may bioaccumulate slightly. Vaughan et al. (1975, cited in Wilber 1980) developed bioconcentration ratios for edible parts of freshwater organisms. Freshwater invertebrates have a bioconcentration factor (BCF) of 10, while freshwater fish have a BCF of 2. Bioaccumulation of Be is not important in determining its aquatic fate (Wilber 1980).

#### 4.5 REFERENCES CITED FOR BERYLLIUM

Duffus, J.H. 1980. Environmental Toxicology. John Wiley & Sons. New York. 164 pp.

Durocher, N.L.. 1969. "Air pollution aspects of beryllium and its compounds." Rpt. No PB 188078. National Technical Information Service. Springfield, Va.

EPA (U.S. Environmental Protection Agency). 1975. "Development document for interim final and proposed effluent limitations guidelines and new source performance standards for the ore mining and dressing industry." Point Source Category, Volume 1. EPA 440/1-75/061. Group II. U.S. Government Printing Office. Washington, D.C.

Goyer, R. A. 1986. "Toxic Effects of Metals", In: Toxicology: the basic science of poisons (C.D. Klaassen, M.O. Amdur, and J. Doull, eds.). Macmillan Publishing Company. New York.

Gregory, A.R. 1964. Effects of Air Pollution on Edible Crops. Department of Environmental Sciences and Engineering. North Carolina University, Chapel Hill.

Oak Ridge National Laboratory: 1978. "Reviews of the Environmental Effects of Pollutants: VI. Beryllium." PB-290 966. EPA Rpt. No. 600/1-78-028. U.S. Dept. of Commerce, National Technical Information Service. Springfield, Va.

Tucker, A. 1972. The Toxic Metals. Ballantine. New York.

Wilber, Charles G. 1980. Beryllium - A potential environmental contaminant. Charles C. Thomas, Springfield, Mo.

## 5.0 CADMIUM

The importance of acclimatization to ambient metal concentrations is well illustrated by cadmium (Cd). Many species may occur in areas where naturally occurring Cd concentrations fall within the range of acute toxicity values derived from laboratory toxicity testing (Eisler 1985). However, sublethal effects of Cd on individual organisms, populations, and communities are also documented.

Heavy metal ratios in native fauna are inconsistent with those of indigenous soil and vegetation, reflecting differences in relative mobilities. For both carnivores and herbivores, Cd is accumulated at rates greater than lead and zinc (Roberts and Johnson 1978) and therefore appears to bioconcentrate.

### 5.1 TERRESTRIAL AND AQUATIC VEGETATION

Cadmium is thought to be one of the most toxic elements for plants. Taylor et al. (1991) introduced wheat to various levels of Cd in soil media and determined the threshold to be 0.02  $\mu\text{M}$ , which translates to a 152 percent growth reduction per  $\mu\text{M}$ . Cadmium was found to be more toxic to wheat than aluminum, copper, manganese, nickel, and zinc.

Adema and Henzen (1989) determined  $\text{EC}_{50}$  values (the concentration at which the weight of the plants is half that of the control plants) and no observed effect concentration (NOEC) values of Cd for growth of lettuce, oats, and tomatoes in loamy soil.  $\text{EC}_{50}$  values for lettuce, oats, and tomatoes were 33, 159, and 171, respectively. NOEC values were 3.2, 10, and 32. Huebert and Shay (1991) determined that the threshold toxicity value for the duckweed *Lemna trisulca* was 116  $\mu\text{g Cd/g soil}$  (oven-dry weight). Values  $>116 \mu\text{g/g}$  caused a reduction in growth. In this study, the  $\text{EC}_{50}$  was 76 parts per billion (ppb) in soil.

The effect and accumulation of Cd in lettuce (*Lactuca sativa*) grown in hydroponic solution was found to be affected by the concentration of other trace elements. Consequently, no absolute toxicity limits for Cd can be drawn without considering other trace elements (Thys et al. 1991).

Calcium, phosphorus, zinc, copper, and manganese reportedly impede Cd uptake. However, the results were not conclusive and seemed to depend on several other factors, such as plant species, and varieties.

OECD (1975) found that Cd caused reductions in yield in eight agronomic plants grown hydroponically. Three-week old plants showed 50 percent growth reduction over a subsequent 19-day period of treatment with Cd ions, as follows: beans, beets, turnips--0.2 mg/l; corn, lettuce--1.0 mg/l; tomatoes, barley--5.0 mg/l; and cabbage--9.0 mg/l.

The Cd content of surface agricultural soils has been found to range from traces to 4.67 mg/kg, with an average of  $0.88 \pm 0.79$  mg/kg in 33 soils. The common natural level for Cd in soils is probably 1 mg/kg (OECD 1975). Cd concentrations in soil above 250 µg/g (dry weight) may cause partial elimination of soil microflora (OECD 1975). Few studies of bioaccumulation and food web dispersal have been conducted involving plants and Cd. Because toxic Cd levels are known for soil microflora, it is possible that plants are affected by poor soil conditions long before Cd levels within the plant can reach toxic concentrations.

## 5.2 TERRESTRIAL INVERTEBRATES

Cd has been shown to have a higher potential for concentration than any other metal in most terrestrial invertebrates. Earthworms have been a focal study organism, due to their soil dwelling and ingesting habits. Other herbivorous or detritivorous invertebrates studied include slugs, snails, and woodlice. Typically, the exposure pathway via food or litter is assumed rather than shown experimentally, although several authors transferred control animals to contaminated soils from sites of concern.

Mortality associated with high levels of Cd in tissues was reported in earthworms (Hartenstein et al. 1981), although the metal did appear to affect metabolic processes and reproduction in snails (Russell et al. 1981) or earthworms (Hartenstein et al. 1981). Acute exposure was seen only in feeding experiments on snails by Berger and Dallinger (1981) and Russell et al. (1981),



neither of whom found excess mortality. Berger and Dallinger (1989) reported maximum concentration of Cd in the midgut of *Arianta arbustorum* (500 µg/g; a concentration factor of 2.2). Most of the Cd intake was retained in tissues until the 8th day when elimination increased to 30 percent; elimination increased further to 45 percent after 20 days. After feeding was discontinued, the ratio between assimilation and loss remained constant. *Helix aspersa* were fed a diet containing concentrations of Cd ranging from 10 to 1000 ppm. As Cd concentration increased, a number of effects were noted: (1) feeding rates decreased, resulting in decreases in weight; (2) the rate of dormancy increased; (3) shell growth declined; and (4) reproductive activity, adduced by observed mating attempts or spermatophores, declined (Russell et al. 1981). Tissue concentrations of Cd rose slowly and leveled off in the 300 ppm treatment; mortality was very low in all treatment groups.

Chronic exposures were assumed to be associated with field-collected animals living at contaminated sites. Many studies involved collecting invertebrates from such sites and characterizing their whole body or tissue-specific Cd burdens. A composite sample of worms of three genera (*Lumbricus*, *Alabophera*, and *Octolasion*) from an unpolluted site showed dry weight Cd concentrations of 3.4 to 9.3 ppm (Van Hook 1974). These values give bioconcentration factors of 11.6 to 22.5 compared to soil, which had a mean Cd concentration of 0.35 ppm. Morgan and Morgan (1990) found the greatest concentration of Cd in the posterior alimentary canal of *Lumbricus rubellus* from a mine site. Here, mean Cd concentration was  $2,639 \pm 398$  µg/g compared to a control value of  $42.3 \pm 4.3$  µg/g, yielding a bioconcentration factor of 62.4. The whole body value was  $811.5 \pm 134.9$  µg/g, yielding a BCF of 66. In many species of earthworms, a linear relationship apparently exists between Cd concentrations in soil and tissue (Martin and Coughtrey 1982).

Composite samples of mixed, unidentified earthworms from a smelter site showed a range of Cd concentrations from 25.2 to 144 µg/g. Individual worms from the same site gave a concentration of 143.7 µg/g. This value represents a BCF of 7.6 over the soil and 4.96 over the litter (Martin and Coughtrey 1976). Soil-feeding and litter-feeding worms from a control site 28 km away had

Cd concentrations of 25.2 µg/g giving bioconcentration factors of 12.6 and 13.6, respectively (Martin and Coughtrey 1975). Wright and Stringer (1980) sampled four species from a smelter site and reported the following Cd concentrations: *Lumbricus terrestris*, 16 - 55 µg/g; *Allobophora calliginosa*, 35 - 63 µg/g; *A. tuberculata*, a maximum of 19 µg/g; *A. chlorotica*, 16 - 55 µg/g; *A. longa*, 17 - 39 µg/g; and *A. rosea*, 17 - 49 µg/g. The BCF from the smelter site (18.8) was much greater than that seen in controls (5.6).

*Lumbricus rubellus* from heavily travelled roadside sites showed relatively low concentrations of Cd (8 µg/g; Weigmann 1991). Weigmann's (1991) value did not differ significantly from the concentration found in control worms from an unpolluted site (9 µg/g). The relatively low Cd content in *L. rubellus* may be due to its preferred habitat deeper in the soil than species (such as *Dendrobaena octoaeda*) found at more surficial levels. *D. octoaeda* from the same roadside site exhibited a BCF of 2.5 (10 µg/g). *Lumbricus terrestris* was found to have a range of concentrations from  $0.55 \pm 0.09$  to  $122.1 \pm 0.28$  µg/g, and *A. chlorotica* from  $0.18 \pm 0.02$  to  $9.30 \pm 0.18$  µg/g.

Ash and Lee (1980) sampled *L. terrestris*, *L. rubellus*, and *A. chlorotica* from roadside sites and found that low levels of Cd were excreted, the bulk concentrated in worm tissues. Cadmium concentrations in worms from contaminated sites ranged from  $3.75 \pm 0.77$  ppm to  $12.10 \pm 0.28$  ppm, which are potentially toxic levels. An earlier study found a similar range of Cd concentrations (4.15 - 12.1 ppm; Ashe and Lee 1979).

In worms used to digest sewage sludge (*Eisenia foetida*), Helmke et al. (1979) found that Cd concentrations in tissues increased with increasing sludge applications; however, Hartenstein et al. (1980) found no such increase. When radioactively labeled Cd was added to the culture, uptake proceeded in a linear fashion while elimination followed first order kinetics (Helmke et al. 1979). The half-life for Cd elimination occurred over two periods ending after  $120 \pm 40$  days. Growth inhibition was seen at Cd concentrations of 1,800 to 18,000 mg/kg and mortality at 3,500 to

35,000 mg/kg (Hartenstein et al. 1981). Interestingly, Cd was one of the least toxic metals with respect to growth inhibition in this study.

Slugs are also relatively sedentary herbivorous invertebrates and have been sampled extensively from polluted sites. Greville and Morgan (1989, 1990) found no consistent patterns in Cd accumulation that were attributable to species or body weight of six species from three genera of slugs (*Deroceras*, *Arion*, and *Milax*) at an old mine site; concentrations ranged from  $34.4 \pm 4.3$  to  $119.3 \pm \mu\text{g/g}$ . Ireland (1979) found a BCF of 10 in *A. ater* at an old mine site; most of the Cd was concentrated in the digestive gland. These animals carried body burdens of  $0.020 \pm 0.004$  milligrams per gram (mg/g), compared to  $0.002 \pm 0.0002$  mg/g for controls. Martin and Coughtrey (1982) examined these slug species (*Arion hortensis*, *A. fasciatus*, and *Agriolimax reticulatus*) from sites with varying contamination and found lower bioconcentration factors from the most contaminated sites (0.75 to 8.18), intermediate ratios from moderately contaminated sites (2.48 to 5.56), and higher ratios from least contaminated sites (2.78 to 8.16).

Greville and Morgan (1991) transferred slugs of two species (*Arion subfuscus* and *Deroceras reticulatum*) from an unpolluted site to contaminated soil. The experimental slugs accumulated Cd until, after 20 days, they matched the concentration of the slugs taken from the mine site; maximum levels attained were  $44.5 \pm 5.5 \mu\text{g/g}$  in *A. subfuscus* and  $53.7 \pm 2.9 \mu\text{g/g}$  in *D. reticulatum*.

Cadmium concentration in slugs from smelter sites ranged from 28.6 to 51.9 ppm, yielding a BCF of 5.28 above vegetation and 1.13 above litter (Martin and Coughtrey 1976). Snails (*Helix aspersa*) taken from the same site were found with Cd concentrations of  $52.45 \pm 19.17 \mu\text{g/g}$  (coefficient of variation 0.37), yielding a BCF of 41.6 above vegetation and 4.46 above litter (Martin and Coughtrey 1976). A significant positive correlation ( $r=0.727$  and  $0.623$ ) between body weight and Cd concentration indicated that body size influenced the metal accumulation process.

Cadmium is apparently also concentrated in snails, which are an important prey for some secondary consumers. Coughtrey and Martin (1977) reported higher concentrations in *Helix aspera* from a contaminated urban site (11.01 µg/g) than previously reported by the same authors for *Oxychilus* spp. and *H. aspera* for uncontaminated sites 7.6 and 6.04 µg/g, respectively (Martin and Coughtrey 1975; Coughtrey and Martin 1976). Bioconcentration factors were higher at heavily contaminated sites (7.57) than at moderately contaminated sites (1.66 - 4.77) and uncontaminated sites (2.32 - 3.64) (Martin and Coughtrey 1982). Interestingly, Williamson (1980) found that Cd concentrations in the snail *Cepaea hortensis* increased with age (as would be predicted) but was inversely correlated with weight.

Woodlice have been sampled for Cd accumulation in a variety of environments. Martin and Coughtrey (1982) suggested that woodlice are ideal indicators of environmental Cd because (1) the relationship between Cd concentration in the animals and the litter is constant over three orders of magnitude and (2) these animals have particularly high concentration factors for Cd. *Oniscus asellus*, collected at different distances from a smelter site, showed a maximum concentration of 202 ppm, indicating a BCF of 5.9 above litter (Martin et al. 1976) and 21.4 above soil (Martin and Coughtrey 1976). Cadmium levels increased more rapidly in more contaminated sites (Coughtrey et al. 1977). Martin et al. (1976) concluded that Cd is released more readily than other metals from the litter, which is the food source for these isopods, resulting in the observed high bioconcentration factors. An alternate source for these high Cd concentrations could be coprophagy; metals released during digestion might be absorbed if feces are ingested. This dietary peculiarity may occur due to the inefficient design of the isopod digestive system: the digestive areas are located posterior to the absorptive area (Wieser 1978).

Coughtrey et al. (1977) reported a strong linear relationship between Cd concentrations in tissue and litter, although the Cd concentration was probably not regulated by *O. asellus*. Cadmium was primarily sequestered in the hepatopancreas, not the exoskeleton as is the case in many invertebrates (Coughtrey et al. 1977). Up to 95.8 percent of the body burden of the metal was

found in this organ, accounting for as much as 0.5 percent of the dry weight of the animal (Hopkin and Martin 1982).

Hopkin and Martin (1982) determined Cd concentrations in *O. asellus* from various sites, including mines and uncontaminated areas. The maximum total body burden of animals from polluted sites was  $140 \pm 5$   $\mu\text{g/g}$ ; in controls, Cd concentration was  $11.5 \pm 1.1$   $\mu\text{g/g}$ . Isopods from polluted sites have greater tolerance for Cd than animals from unpolluted sites (i.e., higher survivorship on polluted litter), although total body burdens are not lower (Martin and Coughtrey 1982). Cadmium concentrations in grasshoppers, sampled at varying distances from a smelter, increased in a nonlinear fashion from 0.4 ppm at 200 km to 7.1 ppm at 2.4 km, giving bioconcentration factors of 1.33 to 3.94 (Munshower 1972).

Studies assessing the movement of Cd through food chains are rare. Roberts and Johnson (1978) analyzed food chain transfer for Cd at a mine site. In a number of groups of herbivorous invertebrates, including grasshoppers, beetles, earwigs, springtails, flies, and ants, the authors found Cd concentrated above the level in vegetation but not significantly above litter values (approximately 11  $\mu\text{g/g}$ ). Cadmium concentration in carnivorous invertebrates including spiders, beetles, and centipedes averaged 34  $\mu\text{g/g}$ , yielding a BCF of 3-4 above the prey groups. Weigmann (1991) found no increase in Cd concentration in carnivorous centipedes despite moderate accumulation in earthworms, a potential prey item.

In a deciduous woodland in eastern Tennessee, Andren et al. (1973) reported moderate levels of Cd in all levels of the ecosystem, although bioconcentration seemed limited to canopy-feeding insects and earthworms. Soil and litter values for Cd were 0.2 and 0.4 ppm, respectively. In plant tissue, Cd concentrations were localized as follows: 0.1 ppm in acorns, 0.2 in branches, and 1 ppm in leaves and roots. In primary consumers, only canopy-feeding insects concentrated the metal, to 5 ppm. Earthworms showed the greatest degree of Cd concentration at 90 ppm. Unfortunately, the authors did not specify feeding relationships in the forest, nor did they identify, beyond broad taxonomic groupings, the consumer organisms.

Van Hook and Yates (1975) found that Cd was not bioaccumulated in a simple grassland food chain consisting of two primary producers (the grasses *Festuca arundinacea* and *Andropogon virginicus*), one primary consumer (the cricket *Acheta domesticus*), and one secondary consumer (the spider *Lycosa* sp.). Concentration ratios in the crickets and spiders were 0.60 and 0.71, respectively. Both the uptake and elimination of Cd were more rapid in the herbivore than the predator.

Lindqvist (1992) analyzed Cd levels in five species of phytophagous insects (three Hymenoptera and two Lepidoptera). Cadmium concentrations in tissue were lower than the other metals studied (copper and zinc). Cadmium concentrations were higher in feces than in the food plant, indicating that although absorption occurred, the metal was eliminated. Unlike the essential metals, zinc and copper, Cd occurred at lower levels in adults than in larvae, possibly due to sequestration in epithelia, which are shed at metamorphosis.

In virtually all invertebrates assessed, Cd exhibits great potential for bioaccumulation. The notable exception is earthworms, presumably due to their dwelling below the level at which the metal was found in the soil (Weigmann 1991). Based on the low observed mortality associated with Cd concentrations exceeding 100 ppm dry weight, it is tempting to speculate that detoxification pathways are protecting these organisms. However, insufficient experimental evidence exists to support this conclusion. Only Russell et al. (1981) attempted to measure changes in significant life history parameters associated with Cd intake; they found decrements in metabolic patterns and reproduction, suggesting that high Cd concentrations do affect fitness.

### 5.3 TERRESTRIAL VERTEBRATES

Wildlife are exposed to Cd primarily via ingestion of contaminated food and drinking water. In some situations, Cd contamination can derive mainly from aerial deposition (Beyer et al. 1985). Contamination is most severe near smelters and urban industrialized areas. Birds and mammals appear to be less sensitive to Cd than are aquatic organisms. Cadmium accumulates with age (Hunter et al. 1981) and is seen at higher concentrations in insectivores such as common shrews

(*Sorex araneus*) than in herbivores such as field voles (*Microtus agrestis*) (Roberts and Johnson 1978; Scanlon 1979).

Sublethal effects of Cd on birds and mammals include reduced growth rate, anemia, hypoplasia in bone marrow and gonads, enlarged heart, and behavioral impacts to adults and progeny. Lowest concentrations of Cd producing significant effects include cardiovascular disease in domestic pigeons (*Columba livia*) exposed to 600 ppb Cd in drinking water and behavioral alterations of progeny after female black ducks (*Anas rubripes*) were fed 4 ppm Cd in their diets (Eisler 1985). Threshold concentrations of dietary Cd having significant physiological effects appear to be around 20 ppm for mallard (*Anas platyrhynchos*) ducklings, with exposures of adult birds ranging up to 75 ppm. Male and female mallards fed 200 ppm dietary Cd survived with no weight loss, but egg production was decreased in females (White and Finley 1978, cited in Eisler 1985).

Bone marrow and hematopoietic effects on rodents are known from dietary exposures of less than 2 ppm (Siewicki et al. 1983). The lowest oral dose causing mortality in laboratory rats and guinea pigs was 250 and 150 mg Cd per kg body weight. A maximum dietary Cd content of 100 µg Cd/kg is recommended to avoid acute toxicity and effects of accumulation of Cd in tissues (EPA 1980, cited in Eisler 1985).

Eisler (1985) points out that EPA criteria for Cd in food for humans (75 µg/day) is probably not protective of wildlife, because birds and wild mammals consume 6 - 7 percent of their body weight per day and thus get a much higher dose than humans, who consume about 1 - 2 percent of their body weight each day.

Cd accumulates in liver and kidneys of vertebrates (Anderson and Van Hook 1973; Johnson et al. 1978). In humans, a Cd concentration of 200 ppm (fresh weight) in renal cortex tissue is the highest level at which no adverse effects are observed. Cadmium associated with liver and kidney of the chipping sparrow (*Spizella passerina*) was eliminated with a half-life of about

100 days. Mallard ducklings fed 20 ppm dietary Cd had accumulated 42 ppm Cd in liver tissue after 12 weeks. Mallards and chickens tolerated 200 ppm Cd in the diet for long periods, producing kidney concentrations of 130 ppm fresh weight.

Gray squirrels (*Sciurus carolinensis*) had higher Cd concentrations in their livers in urban areas (5.96 to 9.11 µg/g) than in rural areas (2.04 to 4.63 µg/g) (McKinnon et al. 1976). High concentrations were seen in kidney and liver tissues of rabbits by a smelting plant, 61 and 5.8 µg/g fresh weight (Gordon 1972). Meadow voles (*Microtus pennsylvanicus*) exposed to sewage sludge containing Cd for four years had fresh weight concentrations of 0.8 to 3.1 mg/kg in their livers and 3.5 to 19.1 mg/kg in their kidneys; in contrast, animals from control fields had 0.1 to 0.7 mg/kg in their livers and 0.3 to 1.1 mg/kg in their kidneys (Maly and Barrett 1984). Cadmium concentrations in the liver and kidney of common shrews, field voles, and wood mice were low (13.6, 20.5; 0.7, 1.7; and 0.4, 2.0 µg/g dry weight, respectively) at control sites, and significantly higher at a copper/cadmium refinery (578, 253; 22.7, 88.5; and 18.2, 41.7 µg/g, respectively) (Hunter et al. 1989).

Whole body concentrations in the vicinity of zinc smelters, in dry weight for carcasses of 10 species of birds, was 2.5 mg/kg downwind and 1.2 mg/kg upwind (Beyer et al. 1985). For mice (*Peromyscus* sp.), values were 2.6 mg/kg downwind and 1.2 mg/kg upwind. For short-tailed shrews (*Blarina* sp.) values were 7.3 mg/kg downwind and 4.8 mg/kg upwind (Beyer et al. 1985). From uncontaminated sites, European starlings had whole body concentrations of 0.05 to 0.24 mg/kg fresh weight.

Cadmium residues in vertebrate kidneys or livers that exceed 10.0 mg/kg fresh weight or 2.0 mg/kg in whole body fresh weight should be considered probable Cd contamination. Levels of 13.0 and 15.0 ppm Cd tissue fresh weight probably represent a significant hazard to animals at higher trophic levels. Residues of 200 ppm fresh weight kidney or more than 5.0 ppm whole animal fresh weight should be considered life-threatening (Eisler 1985).



#### 5.4 AQUATIC FAUNA

Cadmium is toxic to aquatic organisms at relatively low concentrations (<10 ppb). The most bioavailable and toxic form of Cd is  $\text{Cd}^{+2}$ . The prevalence of  $\text{Cd}^{+2}$  in freshwater is dependent on water hardness, pH, sediment chemistry, and dissolved and suspended organic carbon (see Eisler 1985). In general, the higher the hardness, pH, and organic carbon content of water, the more Cd forms inorganic and organic complexes resulting in lower bioavailability. EPA's Ambient Water Quality Criteria for Cd in freshwater is based on hardness of the water in question (EPA 1985). Using these criteria, the state of Colorado has established hardness-dependent standards as follows. The maximum 4-day average concentration of Cd is given by the equation:

$$e^{(0.7852[\ln(\text{hardness})]-2.905)} \quad (\text{in } \mu\text{g/l})$$

where hardness is the concentration of  $\text{CaCO}_3$ . The 1-hour maximum concentration is given by:

$$e^{(1.128[\ln(\text{hardness})]-3.828)} \quad (\text{in } \mu\text{g/l})$$

For water hardness equal to 80, this results in 0.95 and 3.0  $\mu\text{g/l}$  for the four-day and one-hour maxima, respectively. Development of the EPA criteria was based on "acid-soluble" fraction of metals in water. EPA has not officially approved methods for acid extraction of metals and recommends the "total recoverable" metals be used in evaluating attainment of water quality criteria (EPA 1985). This approach probably overestimates the bioavailable  $\text{Cd}^{+2}$  fraction and Cd toxicity, because particulate, inorganic, and organic complexes of Cd are included in the total recoverable fraction.

Mortality resulting from acute exposure to Cd has been measured in several species of aquatic invertebrates and fish. Acute (96 hr)  $\text{LC}_{50}$  values for adult warmwater fish species range from 0.8 to 5 ppb (Eisler 1985). The maximum  $\text{LC}_{10}$  (200 hr) for a similar group of species was 1.5 ppb. Immature and larval fish tended to be slightly more sensitive, but within the same range of concentrations. In standard EPA toxicity tests, the mean acute toxicity value for the cladoceran genus *Daphnia* sp. is approximately 26 ppb. A wide range of sensitivities have been

recorded for aquatic insects. For example, reported  $LC_{50}$  values for larval mayflies range from less than 3 ppb (Spehar 1978) to 2,310 ppb (EPA 1985). In addition to the chemical form of Cd, the sensitivity of invertebrate species to toxicants is heavily dependent upon feeding ecology, life history stage, acclimation to ambient conditions, and available food sources. Eisler (1985) reviewed the literature on Cd toxicity and found that Cd concentrations of <1 ppb to 5 ppb had significant effects on populations of freshwater organisms. The effects included decreased standing crop and growth rate of periphyton species; reduced reproduction and populations of crustaceans, aquatic insects, and fish; and reduced species diversity of an experimental assemblage of benthic macroinvertebrates. More extensive work with marine organisms indicates that Cd toxicity caused reduced growth rates in marine fish and annelids, affected molting in crustaceans, and reduced the life span of the F1 generation of the crab *Pontoporeia affinis* (Sundelin 1983).

Cd accumulates differentially in tissues of freshwater fish. Rainbow trout exposed to 10 ppb Cd accumulated to 1740 ppb in gill, 4900 ppb in liver, and 740 ppb in kidney tissue (Roberts et al. 1979). When exposed to 5 ppb Cd, whole body burdens of Cd in mosquitofish were 2,213 times that of the medium (Giesey et al. 1977). Mollusks also accumulate Cd and many other metals in soft tissues. Oysters exposed to 5 ppb Cd in seawater concentrated Cd 2,000 times. A wide range of bioconcentration factors have been reported for aquatic insects (Enk and Mathias 1977, Giesey et al. 1979, Van Hassel et al. 1980, Eisler 1985). Dipteran families, especially Tipulidae and Chironomidae, exhibit the highest BCFs, with values of 2,000 to 5,000 for Cd exposures less than 10 ppb.

Elimination of Cd depends on the source of the contaminant. Free  $Cd^{+2}$  absorbed from the water column is eliminated more slowly than organically complexed Cd ingested with food. Consequently, bioaccumulation appears to result mostly as a result of bioconcentration from ambient water and not food chain transfer. *Daphnia magna* concentrated Cd only seven-fold from contaminated algae, and fish (*Leucospius delineatus*) fed the daphnids did not concentrate Cd any further. This pattern is generally the case for heavy metal contamination (Rand and

Petrocelli 1985). Eisler (1971, 1985) found that whole body burdens of Cd exceeding 5 ppm in fish represent significant threat to the health of the population. Eisler (1985) suggests that Cd concentrations greater than 10 ppm (fresh weight) or 2 ppm whole body burdens should be considered evidence of Cd contamination and concentrations of 15 ppm (fresh weight) in forage or prey items represent a significant hazard to secondary consumers and predators.

## 5.5 REFERENCES CITED FOR CADMIUM

Adema, D.M.M. and L. Henzen. 1989. "A comparison of plant toxicities of some industrial chemicals in soil culture and soilless culture." Ecotoxicology and Environmental Safety 18: 219-229.

Anderson, S.H. and R.I. Van Hook, Jr. 1973. "Uptake and biological turnover of Cd in chipping sparrows, *Spizella passerina*." Environ. Physio. Biochem. 3:243-247.

Andren, A.W., J.A. Fortescue, G.S. Henderson, D.E. Reichle, and R.I. Hook. 1973. In: Ecology and Analysis of Trace Contaminants - Progress Report June 1972 to January 1973. Pp 61-119. ORNL-NSF-EATC-1, Oak Ridge National Laboratory, Tenn.

Ash, D.D.J. and D. L. Lee. 1980. "Lead, cadmium, copper and iron in earthworms from roadside soil." Env. Poll. 22A:59-68.

Berger, B. and R. Dallinger. 1989. "Accumulation of cadmium and copper by the terrestrial snail *Arianta arbustorum* L.--kinetics and budgets." Oecologia 79:60-65.

Beyer, W.N., O.H. Pattee, L. Sileo, D.J. Hoffman, and B.M. Mulhern. 1985. "Metal contamination in wildlife living near two zinc smelters." Environ. Pollut. 38A:63-86.

Coughtrey, P.J. and Martin, M.H. 1976. "The distribution of lead, zinc, cadmium and copper within the pulmonate mollusc, *Helix aspersa* Muller." Oecologia 23:315-322.

Coughtrey, P.J. and Martin, M.H. 1977. "The uptake of lead, zinc, cadmium, and copper by the pulmonate mollusc, *Helix aspersa* Muller, and its relevance to the monitoring of heavy metal contamination of the environment." Oecologia 27:65-74.

Coughtrey, P.J., Martin, M.H., Young, E.W. 1977. "The woodlouse *Oniscus asellus* as a monitor of environmental cadmium levels." Chemosphere 6:827-832.

Eisler. 1971.

Eisler, R. 1985. "Cadmium hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Dept. of Interior, Fish and Wildlife Service. Biological Report 85(1.2)." Contaminant Hazard Reviews Report No. 2. 46 pp.

EPA (U.S. Environmental Protection Agency). 1985.

Giesey et al. 1977.

Gordon, C.C. 1972. "Effects of air pollution on indigenous animals and vegetation." In Helena Valley, Montana, Area Environmental Pollution Study. EPA. Office of Air Programs Publication No. AP.90. N.C.

Greville, R.W. and A.J. Morgan. 1989. "Concentrations of metals (Cu, Pb, Cd, Zn, Ca) in six species of British terrestrial gastropods near a disused lead and zinc mine." J. Moll. Stud. 55:31-36.

Greville, R.W. and A.J. Morgan. 1990. "The influence of size on the accumulated amounts of metals (Cu, Pb, Cd, Zn and Ca) in six species of slug sampled from a contaminated woodland." Moll. Stud. 56:355-362.

Greville, R.W. and A.J. Morgan. 1991. "A comparison of (Pb, Cd, and Zn) accumulation in terrestrial slugs maintained in microcosms: Evidence for metal tolerance." Env. Poll. 74:115-127.

Hartenstein, R., E.F. Neuhauser, and J. Collier. 1980. "Accumulation of heavy metals in the earthworm *Eisenia foetida*." J. Environ. Qual. 9:23-26.

Hartenstein, R., E.F. Neuhauser, and A. Narahara. 1981. "Effects of heavy metal and other elemental additives to activated sludge on growth of *Eisenia foetida*." J. Environ. Quality. 10:372-376.

Helmke, P.A., W.P. Robarge, R.L. Korotov, P.J. Schomberg. 1979. "Effects of soil-applied sewage sludges on concentrations of elements in earthworms." J. Environ. Qual. 8:322-327.

Hopkin, S.P. and M.H. Martin. 1982. "The distribution of zinc, cadmium, lead and copper within the woodlouse *Oniscus asellus* (Crustacea, Isopoda)." Oecologia 54:227-232.

Huebert, D.B. and J.M. Shay. 1991. "The effect of cadmium and its interaction with external calcium in the submerged aquatic macrophyte *Lemna trisulca* L." Aquatic Toxicology 20(1-2): 57-72.

Hunter, B.A., M.S. Johnson, D.J. Thompson, and H. Holden. 1981. "Age accumulation of copper and cadmium in wild populations of small mammals." in International Conference. Heavy Metals in the Environment. Amsterdam. CEP Consultants Ltd, Edinburgh.

Ink and Mathias. 1977.

Ireland, M.P. 1979. "Distribution of essential and toxic metals in the terrestrial gastropod *Arion ater*." Environ. Pollut. 13:271-278.

Lindqvist, L. 1992. "Accumulation of cadmium, copper, and zinc in five species of phytophagous insects." Env. Ent. 21:160-163.

Johnson, M.S., R.D. Roberts, M. Hutton, and M.J. Inskip. 1978. "Distribution of lead, zinc and cadmium in small mammals from polluted environments." Oikos 30:153-159.

Maly, M.S., and G.W. Barrett. 1984. "Effects of two types of nutrient enrichment on the structure and function of contrasting old-field communities." Am. Midl. Nat. 11:342-357.

Martin, M.H. and P.J. Coughtrey. 1975. "Preliminary observations on the levels of cadmium in a contaminated environment." Chemosphere 4:155-160.

Martin, M.H. and P.J. Coughtrey. 1976. "Comparisons between the levels of lead, zinc and cadmium within a contaminated environment." Chemosphere 1:15-20.

Martin, M.H., P.J. Coughtrey, and E.W. Young. 1976. "Observations on the availability of lead, zinc, cadmium and copper in woodland litter and the uptake of lead, zinc and cadmium by the woodlouse, *Oniscus asellus*." Chemosphere 5: 313-318.

Martin, M.H. and P.J. Coughtrey. 1982. Biological monitoring of heavy metal pollution: Land and air. Applied Science Publishers. London.

McKinnon, J.G., G. L. Hoff, W. J. Bigler, and E.C. Prather. 1976. "Heavy metal concentrations in kidneys of urban grey squirrels." J. Wildlife Diseases 12:367-371.

Morgan, J.E. and A.J. Morgan. 1990. "The distribution of cadmium, copper, lead, zinc and calcium in the tissues of the earthworm *Lumbricus rubellus* sampled from one uncontaminated and four polluted soils." Oecologia 84:559-566.

Munshower. 1972.

OECD (Organization for Economic Cooperation and Development). 1979. "Cadmium and the Environment: Toxicity, Economy, Control." Paris. OECD 89 pp.

Rand, G.M. and S.R. Petrocelli. 1985. Fundamentals of Aquatic Toxicology: Methods and Applications. Hemisphere Publishing Corp., Washington, D.C. 666pp.

Roberts, R.D. and M.S. Johnson. 1978. "Dispersal of heavy metals from abandoned mine workings and their transference through terrestrial food chains." Env. Pollution 16:293-310.

Roberts et al. 1979.

Russell, L.K., J.I. Dehaven, R.P. Botts. 1981. "Toxic effects of cadmium on the garden snail (*Helix aspersa*).\" Bull. Environ. Contam. Toxicol. 26:634-640.

Scanlon, P.R. 1979. "Ecological implications of heavy metal contamination of roadside habitats.\" Proc. Ann. Conf. SE Assoc. Fish and Wildlife Agencies 33:136-145.

Siewicky, T.C., J.E. Balthrop, and J.S. Sydlowski. 1983. "Iron metabolism of mice fed low levels of physiologically bound cadmium in oyster or cadmium chloride.\" J. Nutr. 113:1140-1149.

Spehar. 1978.

Sundelin. 1983.

Taylor, G.J., K.J. Stadt, and M.R.T. Dale. 1991. "Modelling the phytotoxicity of aluminum, cadmium, copper, manganese, nickel, and zinc using the Weibull frequency distribution.\" Canadian Journal of Botany 69(2): 359-367.

Thys, C., P. Vanthomme, E. Screvens, and M. De Proft. 1991. "Interaction of cadmium with zinc, copper, manganese, and iron for lettuce in hydroponic culture.\" Plant Cell and Environment 14 (7): 713-718.

Van Hassel et al. 1980.

Van Hook, R.I. 1974. "Cadmium, lead and zinc distributions between earthworms and soils: Potentials for biological accumulation.\" Bull. Environ. Contam. Toxicol. 12:509-512.

Van Hook and Yates. 1975.

Weigmann, G. 1991. "Heavy metal levels in earthworms of a forest ecosystem influenced by traffic and air pollution.\" Water, Air, Soil Pollut. 57-58:655-663.

Williamson, P. 1980. "Variables affecting body burdens of lead, zinc and cadmium in a roadside population of snails *Cepaea hortensis* Muller.\" Oecologia 44:213-220.

Wilser. 1978.

Wright, M.A. and A. Stringer. 1980. "Lead, zinc and cadmium content of earthworms from pasture in the vicinity of an industrial smelting complex.\" Environ. Pollut. 23A:313-322.

## 6.0 CHROMIUM

Chromium (Cr) is an abundant element in the earth's crust. It occurs in many oxidation states, but only trivalent and hexavalent forms are biologically significant. Chromium in ambient air originates from industrial sources such as ferrochrome production, cement production, ore refining, chemical and refractory processing, and combustion of fossil fuels. Chromium values in air are less than 0.1 microgram per cubic meter ( $\mu\text{g}/\text{m}^3$ ) and range from 0.01 to 0.03  $\mu\text{g}/\text{m}^3$  in industrial areas (Goyer 1986). No biomagnification of Cr has been observed in food chains, and concentrations are usually highest at the lowest trophic levels (Eisler 1986).

### 6.1 TERRESTRIAL AND AQUATIC VEGETATION

Chromium is not presently considered an essential nutrient for plant growth, although it is suspected to be an essential or at least stimulatory element for some species of higher plants, bacteria, and fungi (Arnon 1938). Arnon (1938) reported that Cr might improve the growth of barley plants, at least when combined with nickel and molybdenum in a culture solution. Basically, these findings suggested that Cr, if present in minute quantities, may favorably influence the growth of plants. Rai and Raizada (1988) confirmed an interactive relationship, whether antagonistic or protagonistic, between Cr and other micronutrients. A loss of cellular  $\text{Na}^+$  and  $\text{K}^+$  was found in the blue-green alga *Nostoc muscorum* when Cr concentrations in solution reached 20 micrograms per milliliter ( $\mu\text{g}/\text{ml}$ ).

$\text{EC}_{50}$  and NOEC values were determined by Adema and Henzen (1989) in loamy soil for lettuce, oats, and tomatoes. All plants showed a weight decrease as a symptom of toxic Cr concentrations. Lettuce had an  $\text{EC}_{50}$  value of 1.8 and a NOEC value of 0.35. Oats had an  $\text{EC}_{50}$  value of 7.4 and a NOEC value of 3.5. Tomatoes had an  $\text{EC}_{50}$  value of 6.8 and a NOEC value of 3.2.

Guilizzoni et al. (1984) subjected the aquatic plant *Myriophyllum spicatum* to various Cr concentrations in solution. Reduction in shoot weight and length was observed at concentrations of 50  $\mu\text{g}/\text{l}$ . Chromium content in plant tissue was directly related to solution concentrations.

Heavy metals such as Cr are toxic at very low concentrations to a wide variety of aquatic organisms, and bioconcentration probably occurs (Rawlence and Whitton 1977, Muntau 1981). Huffman and Allaways (1973) report no apparent response in the eighth generation of *Lemna* sp. grown in a dilute nutrient solution containing Cr in concentrations of 2 µg/l. Unfortunately, little information exists on the role of trace heavy metals, including Cr, in bioaccumulation in plants and movement through food chains.

## 6.2 TERRESTRIAL INVERTEBRATES

Data on exposure of invertebrates to Cr is quite limited. In the few invertebrates in which availability and bioaccumulation were examined, Cr was not concentrated. Acute exposure data from earthworms and mosquitoes suggest LOELs, although mortality information from bioassays may overestimate the maximum safe level.

Soni and Abbasi (1981) exposed earthworms (*Pheretima posthuma*) to Cr concentrations of 0-100 ppm in soil. They observed variable mortality and reproduction rates in all but the two highest concentrations of Cr. In the 80 and 100 ppm treatments, 100 percent mortality was seen after 60 days. A trend toward increased reproduction by bits (regeneration from body fragments) was seen at the higher levels of Cr. Abbasi and Soni (1983) exposed the earthworm *Octochaetis pattoni* to Cr from 0-20 ppm in soil and calculated a time dependent LD<sub>50</sub> of 20 ppm at 40 days and 15.14 ppm at 60 days. After 60 days exposure, 25 percent of the worms in the 2 ppm treatment and 70 percent in the 20 ppm treatment had died. Interestingly, reproduction (measured by cocoons, juveniles, and bits) increased with increasing levels of Cr.

In mosquitoes, *Aedes aegypti*, even low Cr concentrations produced mortality and physiological and behavioral alterations (Abbasi et al. 1985, 1988). Pupae placed in water treated with 0.5 and 5 ppm showed 10 percent and 30 percent mortality after 48 hours. All adults hatching from treated flasks were unable to fly; such an effect would certainly increase the mortality of exposed individuals in a natural setting. The 48-hour LC<sub>50</sub> was 12.5 ppm.



Hartenstein et al. (1980) saw no concentration of Cr in earthworms (*Eisenia foetida*) from sewage sludge. Tissue levels of Cr ranged from 1-14 ppm. Andren et al. (1973) reported similar results from unidentified earthworms: concentration factors less than 1. Helmke et al. (1979) found body concentrations of Cr to increase with increasing applications of sludge, but concluded that the accumulated Cr was not bioavailable. No mortality or physiological effects were seen even at the highest levels of applied Cr (46,000 mg/kg) (Hartenstein et al. 1981).

Chromium concentrations in woodlice, *Oniscus asellus* and *Armadillidium* spp., do not exceed Cr concentrations found in the litter used by these animals as a food supply (Coughtrey unpub, reported in Martin and Coughtrey 1982). Extremely low levels of Cr were assimilated in crickets, *Acheta domesticus*, fed radioactively labeled  $^{51}\text{Cr}$  in a bioassay (Van Hook and Crossley 1969). Retention in these animals was primarily due to clearance time in the gut.

These results suggest species typical responses to Cr concentration. In some earthworms, it appears that soil Cr concentrations below 15 ppm may not result in accumulation but may decrease growth or increase mortality. Chromium effects may be more pronounced in soil dwelling earthworms because the metal binds tightly to humus and thus may be more available to these animals. In chronically exposed populations, tolerant individuals would have been selected thus mortality may not be apparent or may be compensated by increased reproduction. In mosquitoes, mortality data alone are insufficient to predict long term population effects, a caveat that is probably applicable to all other taxa.

### 6.3 TERRESTRIAL VERTEBRATES

Chromium is an essential trace element in humans and at least some laboratory animals; data are lacking for wild populations. Adverse effects have been documented at 5.1 and 10.0 mg of  $\text{Cr}^{+6}$  and  $\text{Cr}^{+3}$ , respectively, per kilogram of diet (Eisler 1986). High concentrations of Cr are normally found in RNA, but its role is unknown. Trace quantities are essential for carbohydrate metabolism in mammals as well as insulin action. In humans, a diet lacking Cr can lead to Cr

deficiency (Goyer 1986). Half-life for elimination of Cr from rats is 0.5, 5.9, and 83.4 days (Mertz 1969).

Tissue concentrations of Cr vary geographically and have been documented as high as 7 µg/kg in lungs of persons in New York and Chicago (Schroeder et al. 1962, cited in Goyer 1986).

#### 6.4 AQUATIC FAUNA

Hexavalent Cr was associated with adverse effects in invertebrates of widely separated taxa: reduced survival and fecundity of the cladoceran *Daphnia magna* at a concentration of 10 ppb and exposure for 32 days (EPA 1980); growth inhibition of the protozoan *Chilomonas paramecium* at 1,100-3,000 ppb during exposures of 19-163 hours (Honig et al., 1980); abnormal movement patterns of larvae of the midge *Chironomus tentans* at 100 ppb in 48 hours (Catalan 1982); and a temporary decrease in hemolymph glucose levels in the freshwater prawn *Macrobrachium lamarrei* surviving 1,840 ppb Cr<sup>+6</sup> for 96 hours (Murti et al. 1983, Eisler 1986).

Long-term exposure of rainbow trout for 180 days to high, but environmentally realistic, concentrations of 0.2 ppm Cr<sup>+6</sup> resulted in elevated levels of Cr in kidney (3.5 mg/kg fresh weight), liver (2.0 mg/kg), and muscle (0.6 mg/kg); after 90 days in Cr-free media, Cr levels were 1.6, 1.3, and 0.5, respectively (Calamari et al 1982, Eisler 1986). Sublethal effects were observed in freshwater teleosts following exposure to Cr<sup>+6</sup>. In the snakehead (*Channa punctatus*), enzyme activities were altered in a wide variety of organs and tissues after exposure for 30 days to 2.6 ppm (Sastry and Sunita 1984); the effects became life threatening after exposure for 120 days (Sastry and Tyagi 1982, Sastry and Sunita 1982, 1983). Adverse effects of Cr to sensitive species have been documented at 10.0 µg/l (ppb) of Cr<sup>+6</sup> and 30.0 µg/l of Cr<sup>+3</sup> in freshwater (Eisler 1986).

#### 6.5 REFERENCES CITED FOR CHROMIUM

Abbasi, S.A., P.C. Nipanay, and R. Soni. 1985. "Environmental consequences of the inhibition in hatching of pupae of *Aedes aegypti* by mercury, zinc and chromium - The abnormal toxicity of zinc." Intern. J. Environmental Studies 24:107-114.

Abbasi, S.A., P.C. Nipanay, and R. Soni. 1988. "Studies on environmental management of mercury (II), chromium (VI) and zinc (II) with respect to the impact on some arthropods and protozoans - Toxicity of zinc." Intern. J. Environmental Studies 32:181-187.

Abbasi, S.A. and R. Soni. 1983. "Stress-induced enhancement of reproduction in earthworm *Octochaetus pattoni* exposed to chromium (VI) and mercury (II) - Implications in environmental management." Intern. J. Environmental Studies 22:43-47.

Adema, D.M.M. and L. Henzen. 1989. "A comparison of plant toxicities of some industrial chemicals in soil culture and soilless culture." Ecotoxicology and Environmental Safety 18: 219-229.

Andren, A.W., J.A. Fortescue, G.S. Henderson, D.E. Reichle, and R.I. Hook. 1973. In: Ecology and Analysis of Trace Contaminants-Progress Report, June 1972 to January 1973. ORNL-NSF-EATC-1, Oak Ridge National Laboratory, Tenn.

Arnon, D.I. 1938. "Micronutrient elements in culture solution experiments with higher plants." American Journal of Botany 25:322-325.

Calamari, D., G.F. Gaggino, and G. Pacchetti. 1982. "Toxicokinetics of low levels of Cd, Cr, Ni and their mixture in long-term treatment on *Salmo gairdneri* Rich." Chemosphere 11:59-70.

Catalan, Z.B. 1982. "Behavioral responses of *Chironomus tentans* to stress as determined by an optical fiber monitoring system." Ph.D. thesis, Univ. Michigan, Ann Arbor. 94 pp.

Eisler, R. 1986. Chromium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U.S. Depart. of the Interior, Fish and Wildlife Survey. Biological report 85(1.6), Contaminant Hazard Reviews, Report No.6. 60 pp.

EPA (U.S. Environmental Protection Agency). 1980. Ambient water quality criteria for chromium. U.S. Environ. Protection Agency Rep., 440/5-80-035.

Goyer, R.A. 1986. Toxic effects of metals. In Toxicology: The Basic Science of Poisons (C.D. Klaassen, M.O. Amdur, and J. Doull, eds.). MacMillan Publishing Company, New York.

Guilizzoni, P., Michael S. Adams, and Neil MacGaffey. 1984. "The effect of chromium on growth and photosynthesis of a submersed macrophyte, *Myriophyllum spicatum*." Ecological Bulletins 36:90-96.

Hartenstein, R., E.F. Neuhauser, and J. Collier. 1980. "Accumulation of heavy metals in the earthworm *Eisenia foetida*." J. Environ. Quality 9:23-26.

Hartenstein, R., E.F. Neuhauser, and A. Narahara. 1981. "Effects of heavy metal and other elemental additives to activated sludge on growth of *Eisenia foetida*." J. Environ. Quality 10:372-376.

Helmke, P.A., W.P. Robarge, R.L. Korotov, and P.J. Schomberg. 1979. "Effects of soil-applied sewage sludges on concentrations of elements in earthworms." J. Environ. Quality 8:322-327.

Honig, R.A., M.J. McGinniss, A.L. Buikema, Fr., and J. Cairns, Jr. 1980. "Toxicity tests of aquatic pollutants using *Chilomonas paramecium* Ehrenberg (Flagellata) populations." Bull. Environ. Contam. Toxicol. 25:169-175.

Huffman, E.W.D., Jr. and W. T. Allaways. 1973. "Growth of plants in solution culture containing low levels of chromium." Plant Physiol. 52:72-75.

Jernelov, A., K. Beijer, and L. Soderlund. 1978. "General aspects of toxicology." In: Principles of Ecotoxicology (G.C. Butler, ed.). SCOPE 12. J. Wiley & Sons. New York.

Martin, M.H. and P.J. Coughtrey. 1982. Biological monitoring of heavy metal pollution: land and air. Applied Science Publishers. London.

Mertz, W. 1969. "Chromium occurrence and function in biological systems." Physiol. Rev. 49:163-239.

Muntau, H. 1981. "Heavy metal distribution in the aquatic ecosystem 'Southern Lake Maggiore.' II. Evaluation and trend analysis." Mem. Ist. Ital. Idrobiol. 38:505-529.

Murti et al. 1983.

Rai, L.C. and M. Raizada. 1988. "Impact of chromium and lead on *Nostoc muscorum*: regulation of toxicity by ascorbic acid, glutathione, and sulfur-containing amino acids." Ecotoxicology and Environmental Safety 15:195-205.

Rawlence, D.J. and J.S. Whitton. 1977. "Elements in aquatic macrophytes, water, plankton, and sediments surveyed in three North Island lakes." N.Z. J. Mar. Freshwat. Res. 11:73-93.

Sastry, K.V. and K. Sunita. 1982. "Effect of cadmium and chromium on the intestinal absorption of glucose in the snakehead fish, *Channa punctatus*." Toxicol. Lett. 10:293-296.

Sastry, K.V. and K. Sunita. 1983. "Enzymological and biochemical changes produced by chronic chromium exposure in a teleost fish, *Channa punctatus*." Toxicol. Lett. 16:9-15.

Sastry, K.V. and K. Sunita. 1984. "Chronic toxic effects of chromium in *Channa punctatus*." Environ. Biol. 5:47-52.

Sastry, K.V. and S. Tyagi. 1982. "Toxic effects of chromium in a freshwater teleost fish, *Channa punctatus*." Toxicol. Lett. 11:17-21.

Schroeder et al. 1962.

Soni, R. and A.S. Abbasi. 1981. "Mortality and reproduction in earthworms *Phoretima posthuma* exposed to chromium (VI)." Intern. J. Environmental Studies 17:147-149.

Van Hook, R.I., and D. A. Crossley. 1969. "Assimilation and biological turnover of cesium-134, iodine-131, and chromium-51 in brown crickets, *Acheta domesticus* (L.)." Health Physics 16:463-467.

## 7.0 COPPER

Copper (Cu) is a common trace element and an essential micronutrient necessary for a wide range of metabolic processes (Flemming and Trevors 1989). Copper is widespread and moderately soluble. Its bioavailability depends on a number of factors, such as pH, redox potential, soil and sediment type, water hardness, and organic content. Copper contamination may enter soils and sediments as a result of smelting, mining, industrial activities, domestic waste emission, and the application of fertilizers, sewage sludge, algicides, fungicides, and molluscicides.

### 7.1 TERRESTRIAL AND AQUATIC VEGETATION

All higher plants require Cu for the metabolism of nutrients (Berry 1975). Copper is highly interactive with other elements in soil. In acidic soils, Cu is often out-competed by Al, causing low Cu uptake in plants. The Cu-iron balance seems to be important in preventing Cu deficiency in plants. A relationship has been found between Cu, iron, and molybdenum which suggests that the balance of these ions may be more critical than the absolute amounts taken up by the plants themselves (Donahue et al. 1983).

Taylor et al. (1991) treated wheat with different concentrations of micronutrients in soil media. Growth reduction occurred at the threshold of 3.4  $\mu\text{M}$  for Cu. Copper caused a 19 percent growth reduction per  $\mu\text{M}$  at this threshold.

Rhoads et al. (1989) determined that Cu concentrations of 150 mg/kg at soil pH <6.5 and 330 mg/kg at soil pH >6.5 reduced growth of tomatoes (*Lycopersicon esculentum*). The study confirmed that Cu is more available to plants in acidic soils if there is not an excess of Al present and, therefore, that the threshold for plants in toxic soils is lower.

Lyngby and Brix (1984) subjected eel grass (*Zostera marina*) to various micronutrient concentrations in a seawater solution. At 5  $\mu$ M concentration, growth inhibition, black discoloration, and cellular leaching occurred. The toxicity of the metals used in the study decreased in the order:  $Hg \geq Cu > Cd \geq Zn > Cr(III) > Pb$ .

Copper has low soil mobility, and the potential for Cu accumulation is substantial over a period of time (Rhoads et al. 1989). Copper is strongly adsorbed to cation exchange sites, especially those on humus particles. Several plants, such as *Viscaria alpina*, are Cu tolerant and are able to grow on soils naturally rich in Cu. Apices of the small lateral roots were either dead or abnormal with irregular branching of the roots at extremely high Cu concentrations in the soil of 3,000  $\mu$ g/g. Most of the Cu in the plant was stored in the flower (Hansen and Gullvag 1984). It is not clear whether Cu bioaccumulates in plants due to it being bound in the soil, and what its inter-relationships with other ions in the soil are.

## 7.2 TERRESTRIAL INVERTEBRATES

Copper is an essential metal that functions as a cofactor in many enzymes and as part of active sites in certain proteins and is probably accumulated by terrestrial invertebrates. Because higher concentrations of Cu inhibit metabolism and may cause mutations (Jernelov et al. 1978), organisms probably regulate their Cu balance. Unfortunately, little is known about Cu concentrations in natural habitats. Few studies reviewed as part of this investigation reported toxic effects, except for occasional reports of effects on growth or reproduction. Most authors found Cu to accumulate.

Morgan and Morgan (1990) found no toxic effects on the earthworm *Lumbricus rubellus* of Cu in soil at an old mine site. Total body burden of Cu was  $38.9 \pm 2.4$   $\mu\text{g/g}$  (versus  $13.4 \pm 0.6$   $\mu\text{g/g}$  in controls) and was concentrated in the posterior alimentary canal. Earthworms *L. terrestris*, *L. rubellus*, and *Allolobophora chlorotica* from five sites around the United Kingdom had variable levels of Cu in their tissues ( $0.83 \pm 0.06$  ppm,  $7.5 \pm 2.7$  ppm,  $3.8 \pm 0.8$  ppm,  $0.20 \pm 0.03$  ppm, and  $8.9 \pm 1.25$  ppm) (Ash and Lee 1979). Van Ree (1969) found decreased fecundity in unidentified earthworms from soils contaminated by fertilizer and agricultural wastes.

Earthworms *Eisenia foetida* used to digest sewage sludge effectively concentrate Cu (Helmke et al. 1979, Hartenstein et al. 1980), with body levels ranging from 20 to 150 ppm. Hartenstein et al. (1981) reported growth inhibition at Cu concentrations from 1,100 to 11,000 ppm in sludge. This study reported possible mortality at Cu concentrations above 22,000 ppm. An earlier study, in which the worms were given aged sludge as opposed to fresh sludge, found mortality occurring at Cu concentrations greater than 2,500 ppm (Hartenstein et al. 1979).

Concentration of Cu was also reported from most mollusks. Berger and Dallinger (1989) fed snails (*Arianta arbustorum*) a diet containing Cu and found a bioconcentration factor of 2.4. The Cu (492  $\mu\text{g/g}$ ) was located primarily in the midgut. Low, but constant, levels of Cu were excreted in the feces. The snail *Helix aspersa* from a smelter site showed Cu concentrations of  $86.74 \pm 24.92$   $\mu\text{g/g}$ ; the coefficient of variation was 0.29 (Coughtrey and Martin 1977). Coughtrey and Martin (1976) determined Cu concentrations from *H. aspersa* from sites with varying levels of Cu contamination. The authors found concentrations of  $46.1 \pm 4.04$   $\mu\text{g/g}$  from an uncontaminated site,  $32.7 \pm 7.82$   $\mu\text{g/g}$  from a moderately contaminated site, and  $67.9 \pm 10.03$   $\mu\text{g/g}$  near a smelter site. The Cu in these animals was generally distributed throughout body tissues. Coughtrey and Martin (1977) reported a strong correlation ( $r=0.84$ ) between Cu concentrations and body weight, but also noted that Cu concentrations was closely correlated with concentrations of other metals such as Cd and zinc; thus, it is difficult to elucidate the physiological effects of each metal (Coughtrey and Martin 1976).

Greville and Morgan (1989, 1990) studied six species of slugs from three genera (*Arion*, *Deroceras*, and *Milax*) at an old mine site. Body levels of Cu ranged from  $59.8 \pm 11.5 \mu\text{g/g}$  to  $122.8 \pm 13.7 \mu\text{g/g}$ , with no consistent patterns emerging among species or genera. Ireland (1979) noted a very slight increase in Cu concentrations in slugs (*Arion ater*) from an old mine site relative to control animals ( $0.09 \pm 0.023 \text{ mg/g}$  versus  $0.05 \pm 0.004 \text{ mg/g}$ ).

Tranvik and Eijsackers (1989) bioassayed collembolans on contaminated and control soil and fungi. If both substrate (soil) and food (fungi) were Cu-contaminated, body Cu concentrations reached  $2,770 \mu\text{g/g}$ ; if only food was contaminated, Cu concentrations were  $288 \mu\text{g/g}$ , not significantly different from control levels ( $310 \mu\text{g/g}$ ). Survivorship was the same on contaminated and control soils; desiccation increased mortality on Cu-contaminated soils relative to control soils (50% versus 18%).

Isopods, like mollusks and earthworms, utilize litter for a significant fraction of their diets. Thus, they are likely to accumulate metals deposited by aerial routes, probably the major source of many heavy metals. Weiser et al. (1976) found high Cu concentrations in isopods (*Tracheoniscus rathkei* and *Oniscus asellus*) from Cu mines (115 - 538 ppm) and nearby areas (74 - 460 ppm). These levels represent concentration factors of 6.13 - 9.25.

A close correlation between Cu in isopods (*Tracheoniscus rathkei*) and litter was reported ( $r = 0.98$ ) (Weiser et al. 1977); the relationship between soil and body levels was variable. Copper levels were much higher in animals from sites with high Cu concentrations than sites with low Cu concentrations in soil or litter.

Isopods of three species (*Porcellio scaber*, *P. laevis*, and *Oniscus asellus*) were fed varying amounts of Cu in their diets corresponding to 20, 340, and 5200  $\mu\text{g/g}$  of Cu (Dallinger and Wieser 1977). Higher Cu concentrations resulted in less feeding but increased assimilation of Cu. Animals with either high or low body burdens of Cu were allowed to select a food source from a number of options with varying Cu content. Those with high body levels selected litter



with low Cu concentration while those with low body levels selected litter with either moderate or low Cu concentration (Dallinger 1977). No toxic effects were noted in these isopods.

A few studies examined movement of Cu through terrestrial food chains. Weigmann (1991) found no difference in Cu concentration between worms (*Dendrobaena octaeda* and *Lumbricus rubellus*) from soil near heavily travelled roads (11 - 12 mg/kg) and control sites (11 - 12 mg/kg). Centipedes, assumed to prey on these earthworms and other potentially contaminated animals, did have elevated Cu concentration (35 - 40 mg/kg), suggesting concentration of Cu through trophic mechanisms.

Lindqvist (1992) examined five species of phytophagous insects (three Hymenoptera and two Lepidoptera). Copper concentration was higher in adults than in larvae, and higher in larvae than in their food plants. Larvae, which tend to be voracious feeders, thus represent a major concentration step in terrestrial food chains. This concentration is significant as many secondary consumers, such as birds, make insect larvae a large portion of their diets.

### 7.3 TERRESTRIAL VERTEBRATES

Copper is relatively nontoxic to mammals, and tolerance limits are generally 10- to 100-fold higher than for aquatic fauna. Rabbits, ponies, and pigs can tolerate high levels, 300 to 800 µg/g dry weight feed in their diets, with no toxicosis (Flemming and Trevors 1989). EPA levels acceptable in drinking water are 1.0 mg/l.

At a zinc smelter, high concentrations of Cu were found in short-tailed shrews (*Blarina*) and mice (*Peromyscus*). At this site, very little of the metal measured in the soil was incorporated in the plant foliage; most contamination in biota came from aerial deposition (Beyer et al. 1985).

Background dry weight concentrations of Cu in whole body, liver, and kidney were 7.4, 14.6, and 19.7 µg/g in the field vole and 12.1, 23.7, and 30.7 µg/g in the common shrew, respectively. The granivorous wood mouse (*Apodemus sylvaticus*) had liver and kidney concentrations of 15.8 and

22.3 µg/g. At a Cu refinery, concentrations in liver and kidney were 26.7 and 35.8 µg/g for the common shrew, 14.4 and 18.5 µg/g for the field vole, and 14.6 and 15.2 µg/g for the wood mouse, respectively (Hunter et al. 1989). Thus, the granivorous wood mouse and the herbivorous field vole showed no increase in Cu content with the increased environmental Cu levels. The predatory common shrew did experience significant Cu accumulation, suggesting a potential for bioconcentration.

#### 7.4 AQUATIC FAUNA

Copper occurs in natural waters primarily as the divalent cupric ion in free and complexed forms (Callahan et al. 1979). The cupric ion is highly reactive and forms moderate to strong complexes and precipitates with many inorganic and organic constituents of natural waters, like carbonates and phosphates. Free cupric ions are more toxic than most organic and inorganic Cu complexes, which tend to reduce toxicity attributable to total Cu (Andrew 1976, Borgmann and Ralph 1983). With this in mind, the interpretation of available toxicity data becomes complicated, because the proportion of free cupric ion present is highly variable and is difficult to measure except under carefully controlled laboratory conditions. Usually, data on Cu toxicity are reported using measurements other than total or dissolved Cu.

Copper is toxic to aquatic life at concentrations only slightly higher than those for plants and animals. Copper is known to act at cell surfaces to exert a toxic effect (MacLeod et al. 1967, Lamb and Tollefson 1973). Most of the available tests on the toxicity of Cu to freshwater animals have been conducted with four salmonid (trout) species, fathead and bluntnose minnows, and bluegills. Acute values range from 6.5 µg/l for *Daphnia magna* in hard water to 10,200 µg/l for the bluegill in hard water (Cairns et al. 1978). Several factors are key contributors to the level at which Cu becomes toxic. These factors include water hardness, pH, and total organic carbon (TOC) level. As a general rule, Cu toxicity decreases with increases in alkalinity and TOC.

Chronic toxicity values are available for fifteen freshwater species. Values range from 3.873 µg/l for brook trout to 60.36 µg/l for northern pike (*Esox lucius*). Fish and invertebrate species seem to be about equally sensitive to the chronic toxicity of Cu (EPA 1984).

Protection of animal species appears to offer adequate protection of plants. Bioconcentration factors in fresh water ranged from zero for the bluegill to 2000 for the alga *Chlorella regularis* (EPA 1984). The maximum permissible tissue concentration has not been determined; therefore, a freshwater Final Residue Value can not be calculated for Cu.

## 7.5 REFERENCES CITED FOR COPPER

Andrew, R.W. 1976. "Effects of inorganic complexing on toxicity of copper to *Daphnia magna*." Water Res. 11:309.

Ash, C.P.J. and D.L. Lee. 1980. "Lead, cadmium, copper and iron in earthworms from roadside sites." Env. Poll. 22A:59-67.

Berger, B. and R. Dallinger. 1989. "Accumulation of cadmium and copper by the terrestrial snail *Arianta arbustorum* L.: kinetics and budgets." Oecologia 79:60-65.

Berry, Wade L. 1975. "Dose-response curves for lettuce subjected to acute toxic levels of copper and zinc." In Proceedings from the 50th Annual Hanford Life Sciences Symposium. Richland, Va.

Beyer, W.N., O.H. Pattee, L. Sileo, D.J. Hoffman, and B.M. Mulhern. 1985. "Metal contamination in wildlife living near two zinc smelters." Environ. Pollut. 38A:63-86.

Borgmann, U. and K.M. Ralph. 1983. "Complexation and toxicity of copper and the free metal bioassay technique." Water Res. 17:1697.

Cairns, J., Jr., et al., 1978. "Effects of fluctuating, sublethal applications of heavy metal solutions upon the gill ventilation response of bluegills (*Lepomis macrochirus*)." EPA-600/3-81-003. National Technical Information Service. Springfield, Va.

Callahan, M.A., et al., 1979. "Water-related environmental fate of 129 priority pollutants." Vol. I. EPA-440/4-79-029a. National Technical Information Service, Springfield. V.

Coughtrey, P.J. and Martin, M.H. 1976. "The distribution of lead, zinc, cadmium and copper within the pulmonate mollusc, *Helix aspersa* Muller." Oecologia 23:315-322.

Coughtrey, P.J. and M. H. Martin. 1977. "The uptake of lead, zinc, cadmium, and copper by the pulmonate mollusc, *Helix aspersa* Muller, and its relevance to the monitoring of heavy metal contamination of the environment." Oecologia 27:65-74.

Dallinger, R. 1977. "The flow of copper through a terrestrial food chain. III. Selection of an optimum copper diet by isopods." Oecologia 30:272-276.

Dallinger, R. and W. Wieser. 1977. "The flow of copper through a terrestrial food chain. I. Copper and nutrition in isopods." Oecologia 30:253-264.

Donahue, R.L., R.W. Miller, and J.C. Shickluna. 1983. Soils: an introduction to soils and plant growth, fifth edition. Prentice-Hall, Inc., Englewood Cliffs, N.J.

EPA (U.S. Environmental Protection Agency). 1984. "Ambient Water Quality Criteria for Copper." EPA 440/5/84/031. National Technical Information Service, Springfield. Va.

Flemming, C.A., and J.T. Trevors. 1989. "Copper toxicity and chemistry in the environment: A review." Water, Air, and Soil Pollution 44:143-158.

Greville, R.W. and A.J. Morgan. 1989. "Concentrations of metals (Cu, Pb, Cd, Zn, Ca) in six species of British terrestrial gastropods near a disused lead and zinc mine." J. Moll. Stud. 55:31-36.

Greville, R.W. and A.J. Morgan. 1990. "The influence of size on the accumulated amounts of metals (Cu, Pb, Cd, Zn and Ca) in six species of slug sampled from a contaminated woodland." J. Moll. Stud. 56:355-362.

Hansen, E.A. and B.M. Gullvag. 1984. "Copper tolerance of *Viscaria alpina*." Ecological Bulletin 36:103-106.

Hartenstein, R., E.F. Neuhauser, and J. Collier. 1980. "Accumulation of heavy metals in the earthworm *Eisenia foetida*." J. Environ. Qual. 9:23-26.

Hartenstein, R., E.F. Neuhauser, and A. Narahara. 1981. "Effects of heavy metal and other elemental additives to activated sludge on growth of *Eisenia foetida*." J. Environ. Qual. 10:372-376.

Helmke, P.A., W.P. Robarge, R.L. Korotov, and P.J. Schomberg. 1979. "Effects of soil-applied sewage sludges on concentrations of elements in earthworms." J. Environ. Qual. 8:322-327.

Hopkin, S.P. and M.H. Martin. 1982. "The distribution of zinc, cadmium, lead and copper within the woodlouse *Oniscus asellus* (Crustacea, Isopoda)." Oecologia 54:227-232.

Hunter, B.A., M.S. Johnson, and D.J. Thompson. 1989. "Ecotoxicology of copper and cadmium in a contaminated grassland ecosystem. IV. Tissue distribution and age accumulation in small mammals." J. Applied Ecology 26:89-99.

Ireland, M.P. 1979. "Distribution of essential and toxic metals in the terrestrial gastropod *Arion ater*." Env. Poll. 13:271-278.

Jernelov, A., K. Beijer, L. Soderlund. 1978. "General aspects of toxicology." In: Principles of Ecotoxicology SCOPE 12. (ed. G.C. Butler) John Wiley & Sons, New York.

Lamb, A. and E.L. Tollefson. 1973. Water Res. 7:599.

Lindqvist, L. 1992. "Accumulation of cadmium, copper, and zinc in five species of phytophagous insects." Env. Ent. 21:160-163.

Lyngby, J.E. and H. Brix. 1984. "The uptake of heavy metals in eelgrass *Zostera marina* and their effect on growth." Ecological Bulletin 36:81-89.

MacLeod, R.A., S.C. Kuo, and R. Gelinas. 1967. J. Bacteriol. 93:961.

Martin, M.H. and P.J. Coughtrey. 1982. Biological monitoring of heavy metal pollution. Applied Science Publishers. London.

Morgan, J.E. and A.J. Morgan. 1990. "The distribution of cadmium, copper, lead, zinc and calcium in the tissues of the earthworm *Lumbricus rubellus* sampled from one uncontaminated and four polluted soils." Oecologia 84:559-566.

Rhoads, F.M., S.M. Olson, and A. Manning. 1989. "Copper toxicity in tomato plants." J. Environ. Qual. 18:195-197.

Taylor, G.J., K.J. Stadt, and M.R.T. Dale. 1991. "Modelling the phytotoxicity of aluminum, cadmium, copper, manganese, nickel, and zinc using the Weibull frequency distribution." Canadian Journal of Botany 69(2):359-367.

Tranvik, L. and H. Eijsackers. 1989. "On the advantage of *Folsomia fimetarioides* over *Isotomiella minor* (Collembola) in a metal polluted soil." Oecologia 80:195-200.

Van Ree, J.A. 1969. "Effects of biocides and their residues on earthworms." Meded. Rijksfac. Landbouwentensc Gent. 34:682-689.

Weigmann, G. 1991. "Heavy metal levels in earthworms of a forest ecosystem influenced by traffic and air pollution." Water, Air and Soil Pollut. 57-58:655-663.

Wieser, W., G. Busch, and L. Buchel. 1976. "Isopods as indicators of the copper contents of soil and litter." Oecologia 23:107-114.

Wieser, W., R. Dallinger, and G. Busch. 1977. "The flow of copper through a terrestrial food chain. II. Factors influencing the copper content of isopods." Oecologia 30:265-272.

## 8.0 CYANIDE

Although cyanide (CN) is ubiquitous in the environment, levels tend to be elevated in the vicinity of metal processing operations, electroplaters, gold and other metal-mining facilities, oil refineries, power plants, and solid waste combustion sites. Manufacture of synthetic fabrics and plastics, pesticidal agents, and predator control devices are additional sources. Natural sources of elevated CN levels occur in many food and forage plants (Eisler 1991). Many chemical forms of CN exist in the environment. Free CN (the sum of molecular hydrogen CN, HCN, and the CN anion,  $\text{CN}^{-1}$ ) is the primary toxic agent, regardless of origin.

No reports were found of CN biomagnification or cycling in living organisms, probably owing to its rapid detoxication. Cyanide seldom persists in surface waters and soil owing to complexation or sedimentation, microbial metabolism, and loss from volatilization (Eisler 1991).

### 8.1 TERRESTRIAL AND AQUATIC VEGETATION

In higher plants, elevated cyanide concentrations inhibit respiration and ATP production, and other processes dependent on ATP, eventually leading to death (Towill et al. 1978). At lower concentrations, effects include inhibition of germination and growth, but cyanide may sometimes enhance seed germination (Eisler 1991). Over 1,000 species of plants, including cassava, sorghum, flax, cherries, almonds, and beans, contain elevated levels of cyanogenic glycosides that release HCN when hydrolyzed (Towill et al. 1978, Leduc 1984). Factors favoring CN accumulation in cyanogenic plants include high nitrogen and low phosphorus in soils (Biehl 1984). Cyanogenesis has an important role in plant defense against predatory herbivores. Foliage of the lima bean, a plant with elevated CN content, has up to 31 mg/kg in some varieties (Brattsten et al 1983). Adverse effects of CN on aquatic plants are unlikely at concentrations that cause acute effects to most species of freshwater and marine fishes and invertebrates (EPA 1980).

## 8.2 TERRESTRIAL INVERTEBRATES

Several arthropods contain high levels of CN which seems to function as a predator defense mechanism (Nahrstedt 1988). Cyanide is also used as a feeding stimulant in some insects. Feeding was stimulated in southern army-worms (*Spodoptera eridania*) up to levels of 10,000 mg KCN/kg in the diet if the larvae were exposed incrementally. Previously unexposed larvae showed reversible signs of poisoning at 10,000 mg/kg in the diet; 5,000 mg thiocyanate per kg diet reduced pupation by 77 percent, completely inhibited oviposition, and reduced adult emergence by 80 percent (Brattsten et al. 1983). Data on other terrestrial invertebrates are scarce.

## 8.3 TERRESTRIAL VERTEBRATES

Single large exposures of CN are extremely lethal and have been used in mass suicides and genocides. However, repeated sublethal doses--especially in diets--can be tolerated by many species for extended periods, and perhaps indefinitely.

More than 1,000 species of food plants and forage crops show elevated CN levels under normal conditions; this probably represents the greatest source of CN exposure and toxicosis to humans and range animals. These plants contain CN in the form of cyanogenic glycosides (Eisler 1991).

Adverse nonlethal effects were noted at drinking water concentrations >150 mg HCN/l and at dietary concentrations >720 mg HCN/kg (Eisler 1991). The LD<sub>50</sub> for CN is 1.43 mg/kg body weight (BW) in mallards, 2.54 mg/kg BW in the black vulture (*Coragyps atratus*), and 11.1 mg/kg BW in the domestic chicken (Wiemeyer et al 1986). For mammals, LD<sub>50</sub> values are 4.1 mg/kg BW in the coyote (*Canis latrans*) (Wiemeyer et al. 1986). Domestic fowl and livestock are protected from harmful effects of CN at <100 mg/kg in their diet.

## 8.4 AQUATIC FAUNA

Cyanide acts rapidly in aquatic environments, does not persist for extended periods, and is highly species selective; organisms usually recover quickly on removal to clean water. Gills, egg capsules, and other gaseous exchange sites are most susceptible (Eisler 1991). Fish were the

most sensitive aquatic organisms tested under controlled conditions. Significant adverse nonlethal effects, including reduced swimming performance and inhibited reproduction were observed in the range of 5.0 to 7.2 µg free CN/l; deaths were recorded for most species between 20 and 76 µg/l. Among invertebrates, adverse nonlethal effects were documented between 18 and 43 µg/l, and lethal effects between 30 and 100 µg/l, although some deaths were recorded at much lower concentrations (Eisler 1991).

## 8.5 REFERENCES CITED FOR CYANIDE

Biehl, M. 1984. Cyanide toxicosis. Veterinary Professional Topics, University of Illinois at Urbana, Cooperative Extension Service 10:5-6.

Brattsten, L.B., J.H. Samuelian, K.Y. Long, S.A. Kincaid, and C.K. Evans. 1983. "Cyanide as a feeding stimulant for the southern armyworm, *Spodoptera eridania*." Ecol. Entomol. 8:125-132.

Eisler, R. 1991. "Cyanide Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review." U.S. Dept. of Interior, Fish and Wildlife Service. Biological Report 85(1.23). Contaminant Hazard reviews Report No. 23. 55 pp.

EPA (U.S. Environmental Protection Agency). 1980. Ambient water quality criteria for cyanides. U.S. Environ. Prot. Agency Rep. 440/5-80-037. 72 pp.

Leduc, G. 1984. "Cyanides in water: toxicological significance." Pp. 153-224 in (L.J. Wever, ed.) Aquatic Toxicology Vol. 2, Raven Press. New York.

Nahrstedt, A. 1988. "Cyanogenesis and the role of cyanogenic compounds in insects." Pp. 131-150 in (D. Evered and S. Harnett, eds.) Cyanide compounds in biology. Ciba Found. Symp. 140. John Wiley. Chichester, England.

Towill, L.E., J.S. Drury, B.L. Whitfield, E.B. Lewis, E.L. Galyan, and A.S. Hammons. 1978. Reviews of the environmental effects of pollutants: v. cyanide. U.S. Environ. Prot. Agency Rep. 600/1-78-027. 191 pp.

Wiemeyer, J.S.N., E.F. Hill, J.W. Carpenter, and A.J. Krynitsky. 1986. "Acute oral toxicity of sodium cyanide in birds." J. Wildl. Dis. 22:538-546.

## 9.0 IRON

Iron (Fe) an essential metal and a dietary requirement of organisms in extremely low amounts; higher intakes can be toxic. In terrestrial ecosystems, Fe is typically bound to soil particles.



Increased soil acidity may release potentially toxic amounts for uptake (Jernelov et al. 1978). In many contaminated ecosystems, soil Fe levels may be sufficiently high to warrant concern over uptake by soil dwelling organisms. Another potential source of elevated Fe in terrestrial ecosystems derives from the natural concentration in litter and soils from contaminated vegetation: as the plant biomass decomposes, Fe concentration increases as other components are lost from the ecosystem (Hughes et al. 1980).

## 9.1 TERRESTRIAL AND AQUATIC VEGETATION

As a plant nutrient, iron is essential for the synthesis of chlorophyll, although it is not part of the structure of the molecule. Fe is the center of the porphyrin ring of the cytochromes and, therefore, is involved both in the transformation of radiant energy and the utilization of energy within the cell (Barbour et al. 1987).

Strong acidity in soil and water can result in possible toxicities of Fe to plants (Donahue et al. 1983). Solubility and mobility of Fe are key to plant toxicity because Fe is found in large amounts in soils. Most Fe is not in forms available to plants. Iron oxides, carbonates, phosphates, and hydroxides have quite low solubilities; Fe also bonds to both insoluble and soluble organic chelates. The low-solubility Fe salts and insoluble chelates are not readily available to plants. High levels of bicarbonate and phosphate also lower Fe availability to plants because of the formation of relatively insoluble Fe salts.

As with many other elements, Fe illustrates some interactive relationships with other soil nutrients. Manganese can interfere with Fe uptake and cause Fe deficiencies in macadamia plants (Nagao and Hirae 1992). Plants are generally normal in coloration when leaf Mn is in the range of 100 - 500 mg/kg, soil pH is <6.0, leaf phosphorus is <0.2 percent, and soil mineralogy is able to provide sufficient levels of Fe. Thus, delicate interactions between P, Fe, and Al affect the amount of uptake of any or all of these elements.

Iron toxicity is one of the main constraints to growth in rice (*Oryza sativa*) on acid sulfate soils (Moore et al. 1989). Root rot was detected in rice plants when critical leaf Fe content reached 300 mg/kg. The Fe toxicity in rice can be expected to occur when  $E^{\circ}$ -Fe exceeds 0.75, and is somewhat independent of the  $Fe^{+2}$ .

Iron deficiency is generally of greater concern for agronomic plants than iron toxicity. Little work has been done on bioaccumulation of Fe in plants. Generally, Fe that is available in the soil is taken up quickly by plants, soil microflora, and soil microfauna and utilized as a nutrient. Iron takes so many chemical forms, both soluble and insoluble, that it is difficult to determine when and if it is toxic along a food chain.

## 9.2 TERRESTRIAL INVERTEBRATES

At an abandoned mine site, Ireland (1975) found a high correlation between total and available Fe (38 percent at the mine versus 46 percent at the control site). Iron accounted for the highest concentration of any metal at either site. Free Fe was 18,100 ppm at the mine (14,699 ppm at the control area), while total Fe was 47,580 ppm at the mine and 31,720 ppm in control soil. Despite these high levels of soil Fe, none was detected in tissues of earthworms at either the mine (*Dendrobaena rubida*) or the control area (*Eisenia foetida*). This result suggests the worms either regulate their intake of Fe or lack mechanisms for uptake. Hartenstein et al. (1980) studied uptake of Fe by *E. foetida* maintained on sludge. The authors found that, over four weeks, Fe concentration in the worms increased. Tissue levels accounted for  $1,415 \pm 168$  ppm, while gut levels were  $684 \pm 37$  ppm. These results contradict Ireland (1975) and support ability to uptake Fe.

Ash and Lee (1980) examined Fe levels in the earthworms *Allolobophora chlorotica*, *Lumbricus rubellus*, and *L. terrestris*. The results from their controls were opposite to those reported by Hartenstein et al. (1980): *L. rubellus* had tissue Fe concentration of  $512.5 \pm 118.7$  ppm, while fecal samples contained  $1,127 \pm 134$  ppm Fe after the animals were maintained on substrate containing 1 percent Fe. *L. terrestris* showed no detectible Fe concentration. Iron levels in

earthworms collected from roadsides and industrial areas ranged from  $65.3 \pm 7.81$  to  $238.3 \pm 33.2$  ppm in *L. terrestris* and from  $24.00 \pm 4.03$  to  $318.0 \pm 21.85$  ppm in *A. chlorotica*; the mean value for *L. rubellus* was  $85.00 \pm 19.35$  ppm. Iron was unlike the other heavy metals studied in that tissue concentration was unrelated to atomic weight. For other metals, tissue concentrations increased with atomic weight. This finding supports the idea that the worms actively regulate their Fe intake, probably through excretory mechanisms.

Iron levels in woodlice showed even higher accumulations:  $516 \pm 53$  to  $4,187 \pm 576$  ppm in *Oniscus asellus* (Martin and Coughtrey 1982). Interestingly, the lower end of the range in animals from old lead mines; the higher levels come from isopods from serpentine soils. Concentration ratios in these animals ranged from 0.03 to 0.09.

### 9.3 TERRESTRIAL VERTEBRATES

The effects on humans of chronic exposure to excess Fe is mitigated by a complex mechanism to maintain homeostasis. Approximately 2 - 15 percent is absorbed from the gastrointestinal tract, and only 0.01 percent of absorbed Fe is eliminated per day. Normally, excess ingested Fe is excreted, and some is contained within shed intestinal cells and in bile and urine, and in even smaller amounts in sweat, nails, and hair. Total Fe excretion is usually in the order of 0.5 mg/day. Accidental ingestion can lead to acute toxicity when more than 0.5 g of Fe or 2.5 g of ferrous sulfate are ingested (Goyer 1986).

### 9.4 AQUATIC FAUNA

Iron is an essential trace element for most aquatic organisms and is important in the formation of hemoglobin and myoglobin (NAS 1980). Precipitates of Fe (such as Fe hydroxide) can coat fish gills and inhibit oxygen uptake and can cover sediments and thus suffocate fish eggs and bottom-dwelling organisms (Davies and Goettl 1979, cited in Lehnertz 1989). Ferrous ( $\text{Fe}^{+2}$ ) and ferric ( $\text{Fe}^{+3}$ ) iron are the species of concern in aquatic systems, although ferric iron is practically insoluble (EPA 1986). The Federal chronic ambient water quality criterion for Fe is 1,000  $\mu\text{g/l}$  (EPA 1976, 1986). No acute criterion for Fe has been established by EPA.

According to Smith et al. (1979), the chronic ambient water quality criterion for Fe is too high because of the reported toxicity to aquatic insects at 320 µg/l. An acute (96-hour) LC<sub>50</sub> value of 320 µg/l Fe has been reported for mayflies (*Ephemera subvaria*) at a hardness of 48 mg/l (Warnick and Bell 1969). Stoneflies and caddisflies were also studied and were found to be much less sensitive to Fe than the mayfly. After seven days, at least 50 percent of the insects survived at an exposure concentration of 16 mg/l. Iron concentrations greater than 1900 µg/l were harmful to pike (Doudoroff and Katz 1953). The lowest concentration that was fatal to brook trout within 24 hours was 133,000 µg/l (Doudoroff and Katz 1953). The EPA (1985) reported a chronic value of 9,690 µg/l for brook trout. Little research has been done on Fe bioaccumulation in aquatic systems. Iron generally tends to be in deficit for most plants and animals in terrestrial and aquatic systems.

#### 9.5 REFERENCES CITED FOR IRON

Ash, C.P.J. and D.L. Lee. 1980. "Lead, cadmium, copper and iron in earthworms from roadside sites." Env. Poll. 22A:59-67.

Barbour, M.G., J.H. Burk, and W.D. Pitts. 1987. Terrestrial Plant Ecology, 2nd edition. Benjamin/Cummings Publishing Company, Inc. Menlo Park, CA. 634 pp.

Donahue, R.L., R.W. Miller, and J.C. Shickluna. 1983. Soils: an introduction to soils and plant growth. Prentice-Hall, Inc. Englewood Cliffs, N.J. 667 pp.

Doudoroff, P. and M. Katz. 1953. "Critical review of literature on the toxicity of industrial wastes and their components to fish, II. Metals as salts." Sewage Ind. Wastes 25:802-839. As cited in Rand and Petrocelli, 1985.

EPA (U.S. Environmental Protection Agency). 1976. "Quality criteria for Water. Office of Water Planning and Standards. July, 1976. EPA 440/9-76-023. Washington, D.C.

EPA. 1985. "Environmental Profiles and Hazard Indices for Constituents of Municipal Sludge: Iron." Office of Water Regulations and Standards. June, 1985. Washington, D.C.

EPA. 1986. "Quality Criteria for Water - 1986." Office of Water Regulations and Standards. EPA 440/5-86-001. Washington, D.C. May 1, 1986.

Goyer, R.A. 1986. Toxic effects of metals. In Toxicology: The Basic Science of Poisons (C.D. Klaassen, M.O. Amdur, and J. Doull, eds.). MacMillan Publishing Company, New York.

Hartenstein et al. 1980.

Hughes, M.K., N.W. Lepp, and D.A. Phipps. 1980. "Aerial heavy metal pollution and terrestrial ecosystems." Adv. Ecol. Res. 11:217-327.

Ireland, M.P. 1975. "Metal content of *Dendrobaena rubida* (Oligochaeta) in a base metal mining area." Oikos 26:74-79.

Jernelov, A., K. Beijer, and L. Soderlund. 1978. "General aspects of toxicology." In: Principles of Ecotoxicology. SCOPE 12. (G.C. Butler, ed.). John Wiley & Sons. New York.

Lehnertz, C. 1989. Clear Creek Basin 1989. "The effects of mining on water quality and the aquatic ecosystem." Draft. Prepared for the Colorado Division of Wildlife. Denver, Colo.

Moore, P.A., Jr., and W.H. Patrick, Jr. 1989. "Iron availability and uptake by rice in acid sulfate soils." Soil Sci. Soc. Am. J. 53:471-476.

Martin, M.H. and P.J. Coughtrey. 1982. Biological monitoring of heavy metal pollution. Applied Science Publishers. London. 475 pp.

Nagao, M.A. and H.H. Hirae. 1992. "Macadamia: cultivation and physiology." Critical Reviews in Plant Sciences 10(5):441-470.

National Academy of Sciences. 1980. "Mineral tolerance of domestic animals." Subcommittee on Mineral Toxicity in Animals. National Research Council, Washington, D.C.

Smith, E.J., K.Y. Chen, P.V. Hodson, J.B. Pearce, and D. L. Swanson. 1979. "Iron." Pages 121-125 In: Thurston, R.V., R.C. Russo, C.M. Fetterolf, Jr., T.A. Edsall, and Y.M. Barber, Jr. (eds.). 1979. A review of the EPA Red Book: Quality Criteria for Water. American Fisheries Society. Bethesda, Md. April, 1979.

Warnick, S.L. and H.L. Bell. 1969. "The acute toxicity of some heavy metals to different species of aquatic insects." JWPCF 41(2):280-284.

## 10.0 LEAD

Historically, 98 percent of the lead (Pb) in the biosphere has come from automobile emissions, specifically the combustion of lead alkyl additives in gasoline (Smith 1980). Introduced as a fine aerosol, Pb eventually falls out either in precipitation or in dust onto vegetation and soil. The prevalence of leaded gasoline until recent years has resulted in high accumulations of Pb along

roadsides. Elemental Pb is not taken up through plant roots (Treshow 1978) but in methylated form is bioavailable and more toxic (Klein and Scheunert 1978). The demyelination of axons, harmful in vertebrates, may not be considered significantly toxic to invertebrates. Lead, however, also interferes with the activity of ATP-ase and thus is potentially toxic to all organisms (Jernelov et al. 1978).

## 10.1 TERRESTRIAL AND AQUATIC VEGETATION

Plants may absorb Pb from the soil via roots and from airborne dust on leaf surfaces. Lead inhibits plant growth, reduces photosynthesis, and reduces mitosis and water absorption (Demayo et al. 1982).

For two species of roadside weeds (*Cassia* sp.), pollen germination was reduced by 90 percent and seed germination by 87 percent at Pb levels of about 500 mg/kg dry weight in soil and about 300 mg/kg dry weight in foliage (Krishnayya et al. 1986). Stournaras et al. (1984) had similar findings with a study of soybean (*Glycine max*) cells exposed to Pb. When the cells were exposed to Pb at concentrations of 207 µg/l, growth was inhibited before cells died.

Andersson (1977) concluded that several metals, including Pb, were generally unavailable for plant uptake. Eisler (1988) confirmed that uptake of Pb by terrestrial plants is limited by the low bioavailability of Pb from soils; adverse effects seem to occur only at total concentrations of several hundred mg Pb/kg soil. The EPA (1980) concluded that there is no evidence for biomagnification of Pb in the food chain of vegetation - cattle - dung - dung beetles (Robel et al. 1981), nor is there convincing evidence that terrestrial vegetation is important in food chain biomagnification of Pb.

Although foliar uptake and translocation of Pb nitrate has been demonstrated (Hemphill et al. 1975), foliar uptake of particulate heavy metals is reportedly of minor importance in contributing to the metal concentrations in annual rings (Arvik and Zimdahl 1974). Little (1977) found that more than 90 percent of the heavy metal burden measured for the leaves of deciduous trees was

in the form of surficial deposition that could be removed by washing the leaves in detergent or mild acid solutions (Little 1973).

In aquatic environments, dissolved Pb is the most toxic form. Once on or in plants, Pb enters the food chain (Chow 1970). Jarvis and Jones (1978) concluded that "in situations where additions of heavy metals to soils are likely to increase uptake by plants, the rate at which they are growing therefore becomes an important factor in the movement of heavy metals along the food chain."

## 10.2 TERRESTRIAL INVERTEBRATES

Few studies have examined the accumulation and transference of Pb through food chains. Roberts and Johnson (1978) determined Pb levels in herbivorous (grasshoppers, beetles, earwigs, springtails, bugs, flies, and ants) and carnivorous (spiders, beetles, centipedes) invertebrates at various distances from an old mine. They found relatively low Pb concentrations in the primary consumers (60 µg/g compared to 10 µg/g in controls), with moderate concentration in the secondary (or higher level) consumers (130 µg/g compared to 60 µg/g in controls).

Andren et al. (1973) investigated Pb levels in a variety of trophic levels in a deciduous forest in eastern Tennessee. They found Pb concentrations in the soil and litter to be 5 and 2 ppm, respectively. The metal tended to concentrate in plant tissue: branches, leaves and roots showed the highest levels (4, 7, and 15 ppm, respectively), while acorns had a much lower concentration (0.2 ppm). Canopy feeding insects did not appear to concentrate Pb; composite samples yielded Pb concentrations of 6 ppm. A mixed pattern in bioaccumulation was found for litter-dwelling cryptozoans (6 ppm) and soil-dwelling earthworms (34 ppm). Unfortunately, the authors did not determine taxonomic identity beyond these broad levels, nor did they address feeding relationships in the ecosystem as a whole. Thus, it is difficult to draw conclusions regarding the movement of Pb through the food web.

Weigmann (1991) studied earthworms, and centipedes which may prey on worms, along heavily travelled highways. No toxic effects were noted. Lead concentration in *Dendrobaena octaedra* was 130 mg/kg, significantly greater than that at the control site. Lead concentration in *Lumbricus rubellus* was 45 mg/kg, less than that of controls. The difference between the two species may be due to differences in soil depths they colonized. The bulk of Pb deposition is in the top 5 cm of soil. In the centipede *Lithobius forficatus* Pb concentration was 2.0 to 2.5 mg/kg at both contaminated and control sites. Thus, this predator, if it feeds significantly on earthworms, does not concentrate Pb. In general, earthworms seem to accumulate Pb, although concentration ratios tended to be less than unity. Lead uptake may depend on levels of other ions (specifically Ca in the environment), pH, or other factors. Certainly, tissue concentrations and concentration ratios varied between species and sites.

Roberts and Johnson (1978) found a highly significant correlation between Pb concentration in soil and tissue in earthworms (*Lumbricus terrestris*) at an old mine site. Body burdens also correlated with distance from the mine. No toxic exposures were noted, although the highest concentration was  $146.0 \pm 26.6$   $\mu\text{g/g}$  (controls were  $4.0 \pm 2.1$   $\mu\text{g/g}$ ). Total body burden in *L. rubellus* from an old mine was an unbelievable  $3,667.6 \pm 1,130.3$   $\mu\text{g/g}$  relative to the control level of  $2.2 \pm 0.2$   $\mu\text{g/g}$  (Morgan and Morgan 1990). The bulk of the Pb was concentrated in the posterior alimentary canal; it is possible that some of this astonishing amount was excreted.

Ireland (1975a) also noted high levels of Pb in earthworms at old mine sites. *Dendrobaena rubida* had Pb concentrations of  $4,160 \pm 930$  ppm (control animals were at  $100 \pm 5.0$  ppm); however, both groups exhibited a similar concentration factor for Pb from the soil: 32.8 at the mine and 38.5 at the control site. Animals taken from contaminated soil and placed on clean soil reached control levels of Pb concentration after five days. No toxic effects were seen in animals taken from the control site and moved to soil from the mine site. After 20 days on soils with high Pb levels, the transferred animals still had lower Pb concentration suggesting that some degree of tolerance has developed in the worms native to the mine site. Less than 20 percent of the total body burden of Pb was found in tissues other than intestinal (Ireland 1975b).



Because none of this Pb was present in soluble form, Ireland concluded it was not bioavailable to the worms.

In a study of three species of worms from an old mine site, Morris and Morgan (1986) found large between-species differences in tissue Pb concentration. *Allolobophora calliginosa* had Pb concentration of 2,500 µg/g, *Lumbricus terrestris* had 300 µg/g, and *Octolasion lacteum* had 4,000 µg/g. The authors concluded that the observed body burdens reflected exogenous Pb concentration as well as soil pH (although they found a nonsignificant correlation between soil pH and Pb concentration) and dietary preferences. They also reported that these worms concentrated calcium, not Pb. Because the uptake of Pb is influenced by Ca concentration, species-specific differences in Ca concentration might lead to differences in Pb concentration.

Various earthworm species were sampled at a smelter site; body burden of Pb was 258.1 ppm, reflecting a concentration ratio of 0.34 relative to soil and 0.35 relative to litter (Martin and Coughtrey 1976). Andren et al. (1973) and Van Hook (1974) also reported concentration ratios less than 1.0 for Pb in soils and earthworms.

Wright and Stringer (1980) sampled nine species of *Lumbricus* and *Allolobophora* from a smelter site. They found the average density (for all species) was lower at the smelter (64 worms or 85.5 grams per square meter [g/m<sup>2</sup>]) than at a control site (161.8 worms or 113.7 g/m<sup>2</sup>). The average Pb concentration in worms was 43-83 µg/g at the smelter and 20 - 44 µg/g at the control site. These levels represented concentration factors of 0.30 - 0.57 at the smelter (soil Pb = 147 µg/g) and 0.22 - 0.48 at the control site (soil Pb = 92 µg/g).

Ash and Lee (1980) sampled three species of earthworms (*L. terrestris*, *L. rubellus*, and *A. chlorotica*) from five roadside sites. More Pb was excreted in feces of animals from roadside sites than controls ( $7.10 \pm 0.18$  to  $51.0 \pm 10.66$  ppm versus  $0.75 \pm 0.15$  ppm). Despite higher levels of excretion, Pb accumulated in tissues of worms from roadsides ( $31.60 \pm 0.80$  to  $274.30 \pm 29.9$  ppm) compared to the controls ( $0.96 \pm 0.15$  ppm). The authors noted that these Pb levels

exceeded those known to be toxic to wildfowl. (In humans, levels of Pb are considered high if they exceed 20 ppm.)

Hartenstein et al. (1981) added various metal salts to sewage sludge containing worms (*Eisenia foetida*). They found no effects of Pb on either growth or survival even at the highest doses of 52,000 mg/kg.

Terrestrial mollusks are important detritivores and provide significant amounts of food to birds, small mammals, reptiles, and amphibians. As in earthworms, Pb was detected (sometimes at extremely high levels), but concentration ratios were typically less than one. Despite the maximum levels of Pb reported, very little toxicity was observed.

Greville and Morgan (1990) examined two species of slugs, *Arion subfuscus* and *Deroceras reticulatum*, from an old mine site. They also sampled and transferred slugs from a control site to the polluted mine site. After 20 days on polluted soils, the transferred slugs had higher Pb concentrations than slugs normally resident there ( $214.4 \pm 21.9$  µg/g versus  $116.8 \pm 16.2$  µg/g for *D. reticulatum*;  $462.5 \pm 31.8$  µg/g versus  $377.2 \pm 130.6$  µg/g for *A. subfuscus*).

In six species of slugs from three genera (*Arion*, *Deroceras*, and *Milax*) from an old mine site, Pb concentration ranged from 75 to 300 µg/g (Greville and Morgan 1989, 1990). Ireland (1979) studied *A. ater* from a mine site; Pb concentration was  $0.94 \pm 0.023$  mg/g, versus  $0.005 \pm 0.0009$  mg/g in controls. The author concluded that Pb accumulated but did not concentrate in any specific tissue.

*Cepaea nemoralis* from a zinc mine showed high Pb concentrations of  $365 \pm 65$  µg/g (Coughtrey 1975); surprisingly, body burdens of Pb in this species were much lower in animals from roadside sites (55 µg/g) (Williamson 1980). Williamson (1980) also found that body weight decreased as Pb concentration increased; body weight accounted for 42.8 percent of the variance in Pb concentration. Most of the Pb in these animals was found in the digestive gland. Lead

concentration decreased during the time these snails were maintained on clean substrate, suggesting a clearance mechanism.

Martin and Coughtrey (1976) looked at concentration ratios for slugs *Arion hortensis*, *A. fasciatus*, *Agriolimax reticulatus*, and *Clausilia bidentata* from a smelter site. Lead concentrations were 60.4, 112.4, 88.2, and 208.4 ppm respectively, yielding concentration ratios of 0.42 relative to soil and vegetation and 0.13 relative to litter. Samples of the snail *Helix aspersa* from the same site showed a Pb concentration of 27.6, giving a concentration ratio of 0.43 relative to soil or vegetation and 0.07 relative to litter. The maximum concentration of Pb in these snails was  $38.99 \mu\text{g/g} \pm 20.59$  with a coefficient of variation of 0.53 (Coughtrey and Martin 1977). Concentration ratios for *H. aspersa* from uncontaminated sites were 0.015 - 0.52, from moderately contaminated sites 0.038 - 0.10, and at more contaminated sites were 0.32 - 0.45 (Coughtrey and Martin 1976). Slugs from a smelter site showed a slightly different pattern in concentration factors: 0.09 - 0.42 from the most contaminated sites; 0.016 - 0.39 from moderate contamination; and 0.001 - 0.25 from least contaminated sites (Martin and Coughtrey 1982).

Woodlice are also important elements of the detritivore food chain. As they reside in the upper soil horizon, it is likely that they are exposed to significant amounts of Pb. In general, Pb accumulation and concentration follow the same patterns in isopods as in earthworms and mollusks (accumulation occurs, but concentration factors are low). However, Martin and Coughtrey (1982) concluded that there is no simple relationship between Pb concentration in isopods and in litter. Lead concentration in tissue and litter increase concomitantly, but, at least in *Oniscus asellus*, there is probably little regulation of Pb. Martin and Coughtrey (1976) reported a Pb concentration of 297.3 ppm in *Oniscus asellus*, giving a concentration ratio of 1.43 above soil or vegetation, although the more relevant ratio is 0.42 relative to litter. Hopkin and Martin (1982) reported that the bulk of stored Pb was localized in the hepatopancreas, up to 2.5 percent of the dry weight of the animal. In many invertebrates, Pb is sequestered in the exoskeleton, providing some degree of isolation from the metabolic activities of the animal. *O. asellus* from a Pb mine had body burdens of Pb of  $464 \pm 28 \mu\text{g/g}$  (controls had  $13.5 \pm 1.6 \mu\text{g/g}$ ).

The hepatopancreas of these animals had  $7,474 \pm 704$  and  $274 \pm 38$   $\mu\text{g/g}$  for mine and control sites, respectively. Despite the high levels of Pb in the animals from the mine site, no ill effects were noted.

As in earthworms, some relationship between lead and calcium uptake was found in isopods (Beeby 1978). Increasing the amount of Ca in the diet increased the uptake of Pb in *Porcellio scaber* (Beeby 1978). In both *O. asellus* and *P. scaber*, Pb and Ca correlated highly with body weight and each other, suggesting that the relevant variable in Pb concentration is Ca concentration, not body weight.

In collembolans, also important members of the detritivore food chain, body burdens of Pb increased when fed a diet treated with high concentrations of Pb. Joose and Buker (1979) fed *Orchesella cincta* algae treated with 11,270 or 13,000 ppm Pb. The total body burden of the experimental animals was 130 and 247 ppm, respectively (controls were 10 - 27 ppm). Pb concentration in the feces was 16,500 and 19,670 in the treated groups, indicating that some 44 percent of the ingested Pb was excreted. Despite these high levels of dietary Pb, no mortality was observed in the treated group.

### 10.3 TERRESTRIAL VERTEBRATES

The toxicity of Pb to mammalian systems is widely recognized. Much of the toxicity to vertebrates probably stems from its tendency to demyelinate axons. Toxic concentrations of Pb in vertebrates are mostly due to the ingestion of lead shot. More than a million ducks and geese die annually as a result of such ingestion (Clemens et al. 1975, cited in Eisler 1988). As with other biota, bioaccumulation is also the result of exposures to combustion of leaded gasoline in vehicles. Raptors, in turn, ingest Pb from dead or crippled game, from Pb-poisoned waterfowl that had ingested lead shot, and from roadside mammals and invertebrates that had high exposures. High Pb doses induce abortion, reduce or terminate pregnancy, result in stillbirths, or increase skeletal malformations. Lead toxicosis has been studied mostly in livestock and laboratory animals. Survival was reduced under the following regimens: acute oral doses of

5 mg/kg body weight in rats, chronic oral doses of 0.3 mg/kg body weight in dogs, and dietary levels of 1.7 mg/kg body weight in horses (Eisler 1988).

Although ingestion of food containing biologically incorporated Pb is unlikely in itself to cause Pb poisoning (Stendell 1980, Custer 1984, Pattee 1984; all cited in Eisler 1988), the effects of lower exposure levels are not well known (Nriagu 1978). While the use of Pb arsenate as an insecticide in orchards has decreased, residues remain in upper soil surfaces and will be bioavailable almost indefinitely (Gilmartin et al. 1985, cited in Eisler 1988). Sublethal effects such as a delayed impairment of learning and abnormal social behavior were seen in monkeys administered 0.1 mg Pb/kg BW daily or fed diets containing 0.5 mg Pb/kg.

Differences in response to Pb contamination has been documented to differ based on species, age, season, geographic location, habitat, and the form in which the metal was ingested (Finley and Dieter 1978, Mudge 1983, Srebocan and Rattner 1988; all cited in Eisler 1988). Comparisons at different traffic densities found concentrations of Pb to be lowest in granivores, intermediate in herbivores, and highest in insectivores (Williamson and Evans 1972). Organic lead has much greater impact than inorganic Pb compounds.

Concentrations of Pb in tissues in pigeons were highest in urban areas (Tansy and Roth 1970, Hutton and Goodman 1980) and close to highways (Getz et al. 1979). Starlings had whole body (less skin, bill, and wings) concentrations of 1.088 µg/g in urban areas and 0.681 µg/g in rural areas. Four bird species had higher Pb concentrations near a steel factory (27 µg/g) than farther from the factory (2.5 µg/g) (Dmowski and Karolewski 1979). Songbirds near zinc smelters had 56 ppm dry weight of Pb, and shrews had even higher concentrations. Two cuckoos from the same contaminated area had liver concentrations of 18 and 25 ppm, and appeared healthy (Beyer et al. 1985). In contrast, death resulted from Pb poisoning at liver concentrations of 23 - 38 mg/kg fresh weight in raptors (Pattee et al. 1981, cited in Eisler 1988).

The highest concentrations of Pb in kidney and liver tissues of mice near smelting plants was 110 and 23 µg/g, respectively (Gordon 1972). Shrews had even higher concentrations (110 ppm dry weight) than mice (17 ppm) near a zinc smelter. Kidney concentrations of Pb for the shrews were 280 ppm wet weight, and this was considered to be toxic (Beyer et al. 1985). Livers of horses whose death was a result of Pb contamination contained 5.7 and 4.4 µg/g, and kidneys had 6.5 and 4.8 µg/g. In humans, Pb levels of 20 ppm are considered high.

#### 10.4 AQUATIC FAUNA

Lead adversely affects survival, growth, reproduction, development, and metabolism of most species under controlled conditions, but its effects are substantially modified by numerous physical, chemical, and biological variables (Eisler 1988). In aquatic environments, dissolved Pb was the most toxic form. Effects of Pb toxicity on aquatic organisms were pronounced at elevated water temperatures, reduced pH, in younger life stages, after long exposures, and when organic Pb compounds were present (Eisler 1988).

Adverse effects were noted on *Daphnid magna* reproduction at 1.0 µg Pb<sup>+2</sup>/l. The exposure duration was 19 days and the reproductive impairment affected 10 percent of the study population (Eisler 1988). At concentrations of 10 µg Pb/l, 50 percent of the study population of *D. magna* showed reproductive impairment.

Rainbow trout survival diminished at 3.5 µg of tetraethyllead per liter. The exposure duration for this experiment was 72 hours. An LC<sub>50</sub> was reached at the above concentration (Eisler 1988). Fathead minnows were not as sensitive to Pb as rainbow trout. An LC<sub>50</sub> was reached in 96 hours at a concentration of 6,500 µg Pb<sup>+2</sup>/l (Eisler 1988).

Although Pb is concentrated by biota from water, there is no convincing evidence that it is transferred through food chains (Wong et al. 1978, EPA 1979, Branica and Konrad 1980, Settle and Patterson 1980; all cited in Eisler 1988). In fact, Pb concentrations tended to decrease markedly with increasing trophic level in both detritus-based and grazing aquatic food chains

(Wong et al. 1978, cited in Eisler 1988). In the freshwater food chain of an alga (*Selenastrum capricornutum*), to a daphnid (*Daphnia magna*), to the guppy (*Poecilia reticulata*), Pb accumulation progressively decreased from the alga to the guppy (Vighi 1981, cited in Eisler 1988).

#### 10.5 REFERENCES CITED FOR LEAD

Andersson, A. 1977. "Some aspects of the significance of heavy metals in sewage sludge and related products used as fertilizers." Swed. J. Agric. Res. 7:1-5.

Andren, A.W., J.A.C. Fortescue, G.S. Henderson, D.E. Reichle, and R.I. Hook. 1973. In: Ecology and Analysis of Trace Contaminants - Progress Report, June 1972 to January 1973. ORNL-NSF-EATC-1, Oak Ridge National Laboratory, Tenn.

Arvik, J.H. and R.L. Zimdahl. 1974. "Barriers to the foliar uptake of lead." J. Environ. Qual. 3:369-373.

Ash, D.D.J. and D. L. Lee. (1980). "Lead, cadmium, copper and iron in earthworms from roadside soil." Env. Poll. 22A:59-68.

Beeby, A. 1978. "Interaction of lead and calcium uptake by the woodlouse, *Porcellio scaber* (Isopoda, Porcellionidae)." Oecologia 32:225-262.

Beyer, W.N., O.H. Pattee, L. Sileo, D.J. Hoffman, and B.M. Mulhern. 1985. "Metal contamination in wildlife living near two zinc smelters." Environ. Pollut. 38A:63-86.

Chow. 1970.

Coughtrey. 1975.

Coughtrey, P.J. and M.H. Martin. 1976. "The distribution of lead, zinc, cadmium and copper within the pulmonate mollusc, *Helix aspersa* Muller." Oecologia 23:315-322.

Coughtrey, P.J. and M.H. Martin. 1977. "The uptake of lead, zinc, cadmium, and copper by the pulmonate mollusc, *Helix aspersa* Muller, and its relevance to the monitoring of heavy metal contamination of the environment." Oecologia 27:65-74.

Demayo, A., M.C. Taylor, K.W. Taylor, and P.V. Hodson. 1982. Toxic effects of lead and lead compounds on human health, aquatic life, wildlife, plants, and livestock. CRC Crit. Rev. Environ. Control 12:257-305.

Dmowski, K., and M.A. Karolewski. 1979. "Cumulation of zinc, cadmium and lead in invertebrates and in some vertebrates according to the degree of an area contamination." Ekologia Polska 27:333-349.

Eisler, R. 1988. "Lead hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Dept. of Interior, Fish and Wildlife Service." Biological Report 85 (1.14). Contaminant Hazards Review Report No. 14. 134 pp.

EPA (U.S. Environmental Protection Agency). 1980.

Getz, L.L., G.L. Rolfe, A.W. Haney, R.L. Wortman, R.W. Larimore, J.W. McNurney, J.L. Hudson, H.V. Leland, R.L. Solomon, K.A. Reinbold, and P.W. Price. 1979. "Transport and distribution in a watershed ecosystem." In: Lead in the Environment. (W.R. Boggess and G.G. Wixson, eds.) Ch. 6, pp. 104-134. Castlehouse Publications Ltd for National Science Foundation. Washington, D.C.

Gordon, C.C. 1972. "Effects of air pollution on indigenous animals and vegetation." In: Helena Valley, Montana, Area Environmental Pollution Study. Pp. 95-112. EPA. Office of Air Programs Publication No. AP.90. N.C.

Greville, R.W. and A.J. Morgan. 1989. "Concentrations of metals (Cu, Pb, Cd, Zn, Ca) in six species of British terrestrial gastropods near a disused lead and zinc mine." J. Moll. Stud. 55:31-36.

Greville, R.W. and A.J. Morgan. 1990. "The influence of size on the accumulated amounts of metals (Cu, Pb, Cd, Zn and Ca) in six species of slug sampled from a contaminated woodland." J. Moll. Stud. 56:355-362.

Greville, R.W. and A.J. Morgan. 1991. "A comparison of (Pb, Cd and Zn) accumulation in terrestrial slugs maintained in microcosms: Evidence for metal tolerance." Env. Poll. 74:115-127.

Hartenstein, R., E.F. Neuhauser, and A. Narahara. 1981. "Effects of heavy metal and other elemental additives to activated sludge on growth of *Eisenia foetida*." J. Environ. Qual. 10:372-376.

Hemphill, D.D. and J.H. Rule. 1975. "Foliar uptake and translocation of Pb-210 and Cd-109 by plants." In: International Conference on Heavy Metals in the Environment. Volume 2, Part 1, pp. 77-86. CEP Consultants. Edinburgh.

Hopkin, S.P. and M.H. Martin. 1982. "The distribution of zinc, cadmium, lead and copper within the woodlouse *Oniscus asellus* (Crustacea, Isopoda)." Oecologia 54:227-232.

Hutton and Goodman. 1980.



Ireland, M.P. 1975a. "Metal content of *Dendrobaena rubida* (Oligochaeta) in a base metal mining area." Oikos 26:74-79.

Ireland, M.P. 1975b. "Distribution of lead, zinc and calcium in *Dendrobaena rubida* (Oligochaeta) living in soil contaminated by base metal mining in Wales." Comp. Biochem. Physiol. 52B:551-555.

Ireland, M.P. 1979. "Distribution of essential and toxic metals in the terrestrial gastropod *Arion ater*. Environ." Pollut. 13:271-278.

Jarvis, S.C. and L.H.P. Jones. 1978. "Uptake and transport of cadmium by perennial ryegrass from flowing solution culture with constant concentration of cadmium." Plant and Soil 49:333-342.

Jernelov, A., K. Beijer, and L. Soderlund. 1978. "General aspects of toxicology." In: Principles of Ecotoxicology SCOPE 12. (G.C. Butler, ed.) John Wiley & Sons. New York.

Joose, E.N.G. and J.B. Buker. 1979. "Uptake and excretion of lead by litter-dwelling collembola." Env. Poll. 18:235-2240.

Klein, W. and I. Scheunert. 1978. "Biotic processes." In: Principles of Ecotoxicology. SCOPE 12. (G.C. Butler, ed.) John Wiley & Sons. New York.

Krishnayga et al. 1986.

Little, P. 1973. "A study of heavy metal contamination of leaf surfaces." Environ. Pollut. 5:159-172.

Little. 1977.

Martin, M.H. and P.J. Coughtrey. 1976. "Comparisons between the levels of lead, zinc and cadmium within a contaminated environment." Chemosphere 1:15-20.

Martin, M.H. and P.J. Coughtrey. 1982. Biological monitoring of heavy metal pollution. Applied Science Publishers. London. 475 pp.

Martin, M.H., P.J. Coughtrey, and E.W. Young. 1976. "Observations on the availability of lead, zinc, cadmium and copper in woodland litter and the uptake of lead, zinc and cadmium by the woodlouse, *Oniscus asellus*." Chemosphere 5:313-318.

Morris, B. and A.J. Morgan. 1986. "Calcium-lead interactions in earthworms: Observations on *Lumbricus terrestris* L. sampled from a calcareous abandoned leadmine site." Bull. Environ. Contam. Toxicol. 37:226-233.

Morgan, J.E. and A.J. Morgan. 1990. "The distribution of cadmium, copper, lead, zinc and calcium in the tissues of the earthworm *Lumbricus rubellus* sampled from one uncontaminated and four polluted soils." Oecologia 84:559-566.

Nriagu, J.O (ed.). 1978. "The biogeochemistry of lead in the environment." Part A: Ecological Cycles. Part B: Biological Effects. Elsevier/North Holland Biomedical Press. Amsterdam, Netherlands.

Robel et al. 1981.

Roberts, R.D. and M.S. Johnson. 1978. "Dispersal of heavy metals from abandoned mine workings and their transference through terrestrial food chains." Env. Pollution 16:293-310.

Smith, R.L. 1980. Ecology and Field Biology, 3rd edition. Harper and Row. New York. 835 pp.

Stournaras et al. 1984.

Tansy, M.F., and R.P. Roth. 1970. "Pigeons: A new role in air pollution." J. Air Pollut. Control. Assoc. 20:307-309.

Treshow, M. 1978. "Terrestrial plants and plant communities." In: Principles of Ecotoxicology. SCOPE 12. (G.C. Butler, ed.) John Wiley & Sons. New York.

Van Hook, R.I. 1974. "Cadmium, lead and zinc distributions between earthworms and soils: Potentials for biological accumulation." Bull. Environ. Contam. Toxicol. 12:509-512.

Weigmann, G. 1991. "Heavy metal levels in earthworms of a forest ecosystem influenced by traffic and air pollution." Water, Air, Soil Pollut. 57-58:655-663.

Williamson, P. 1980. "Variables affecting body burdens of lead, zinc and cadmium in a roadside population of snails *Cepaea hortensis* Muller." Oecologia 44:213-220.

Williamson, P., and P.R. Evans. 1972. "Lead levels in roadside invertebrates and small mammals." Bull. Environ. Contam. Toxicol. 8:280-288.

Wright, M.A. and A. Stringer. 1980. "Lead, zinc and cadmium content of earthworms from pasture in the vicinity of an industrial smelting complex." Environ. Pollut. 23A:313-322.

## 11.0 MANGANESE

As an essential element for both plants and animals, manganese (Mn) might be expected to concentrate in biomass. In ecosystems, mobilization of Mn, like that of many other metals,

increases with pH decrements. Potentially toxic concentrations might result; toxicity in humans is well documented. Manganese also has the potential to exert mutagenic effects via its ability to complex with DNA nucleotides.

### 11.1 TERRESTRIAL AND AQUATIC VEGETATION

Manganese is classified as a micronutrient or trace element. Its concentrations are considered adequate for growth in higher plants when it reaches 50 ppm in dry tissue (Salisbury and Ross 1985). Manganese acts as an enzyme catalyst in plants (Barbour et al. 1987).

Sands have few Mn-bearing minerals, calcareous soils precipitate insoluble Mn dioxide, and large amounts of humus in the soil decrease Mn availability, probably by forming insoluble organic complexes or chelates (Donahue et al. 1983). Manganese is relatively insoluble in basic soils but may be so soluble in strongly acid soils that it is toxic (NAS 1973). Manganese toxicity in soybeans was associated with a water-soluble Mn content of 2.5 ppm in an acid soil during a prolonged wet period (NAS 1973). In nutrient cultures, a Mn content as low as 0.5 ppm was toxic to Atlas 46 barley (NAS 1973). Adding lime to acid soils reduces Mn toxicity.

Hue (1988) conducted a study applying low-Mn sewage sludge onto three Hawaii soils (pH 5.0). Lettuce had reduced yields and high plant-Mn concentrations. Manganese phytotoxicity unexpectedly occurred in all sludge-amended treatments of one soil (0, 20, 40, and 80 grams per kilogram [g/kg]) and in the 80 g/kg rate of the soil. The phytotoxic levels of Mn in soil solutions was explained by the Mn complexation by organic ligands creating soluble Mn for plant uptake. Taylor et al. (1991) determined that the threshold for Mn toxicity for wheat was at a concentration of 37  $\mu$ M in soil. This resulted in a 0.2 percent growth reduction  $\mu$ M.

Some plants, such as macadamias, can accumulate large concentrations of Mn that would normally be toxic to other plants. Plants growing in manganiferous soils can accumulate up to 2,500 mg/kg Mn in recently matured leaves (Nagao and Hirae 1992). Leaf Mn increases as soil phosphorus increases. Uptake of Mn is inhibited by aluminum in aluminous soils. As Mn in the

soil solution increases, leaf Mn also increases. This response relationship seems to be related to interactions between phosphorus, iron, and aluminum in the soil substrate (Nagao and Hirae 1992). It is not clear whether Mn is bioaccumulated by plants and transported through the food chain at a toxic level. As with most micronutrients, Mn is interdependent on other elements for availability and solubility in the soil substrate.

## 11.2 TERRESTRIAL INVERTEBRATES

In earthworms, body weight decreased as Mn concentration in sewage sludge increased; however, no toxic effects (either mortality or growth decrements) were seen even at high levels (22,000 mg/kg). Similar levels were observed in worms from activated sludge and control animals (36 ppm and 32 ppm, respectively).

In mollusks, accumulation of Mn showed a similar pattern to zinc or Cu; no toxic effects were reported. In the slug *Arion ater*, Mn was concentrated in the epidermal regions. Slugs from polluted areas had approximately 17 times the Mn concentration as control animals ( $7.48 \pm 0.48$  mg versus  $0.43 \pm 0.09$  mg). In the snail *Helix aspersa* Mn concentration of  $742 \pm 104$   $\mu$ g/g was reported, mainly from the digestive gland.

## 11.3 TERRESTRIAL VERTEBRATES

In humans, the principal route of excretion of Mn is in the feces. Large doses of Mn salts cause gastrointestinal irritation, and systemic toxicity does not seem to result following oral administration of Mn (Goyer 1986). Scant information was found on toxicity and effects of Mn in terrestrial vertebrates. There is no evidence of biomagnification.

## 11.4 AQUATIC FAUNA

Lewis et al. (1979) reported that manganese toxicity can be affected by hardness and pH in aquatic systems. They also reported antagonism of Mn with nickel toxicity, as well as synergistic effects with some other metals. Unfortunately, limited information is available on the toxicity of Mn to freshwater aquatic organisms. No ambient water quality criteria for the protection of

freshwater aquatic life have been established for Mn by EPA. Reported tolerance values indicate that 1.0 mg/l is protective of freshwater species (McKee and Wolf 1963 cited in EPA 1976; Davies and Goettl 1977 cited in Lewis 1979). Dawson (1974, cited in Lewis 1979) has developed a criterion of 0.1 mg/l.

Acute toxicity for eels has been reported at concentrations of 2.2 to 4.1 mg/l over a time period of 8 to 18 hours. A 24-hour  $LC_{50}$  was determined at a concentration of 6045 mg/l for *Orizias* sp. (McKee and Wolf 1963, cited in Lewis 1979). England and Cummings (1971, cited in Lewis 1979) reported a 96-hour  $LC_{50}$  for Mn in young rainbow trout of 16 mg/l. Davies (1980) reported that the acute toxicity of Mn decreases with increased hardness, as well as increased fish size.

Little information was found concerning Mn bioaccumulation in a food chain or food web. Davies (1980) reported that the relationship between size and toxic effects is pronounced. When smaller fish are consumed by larger fish, the impact of toxic effects decreases. Therefore, Mn does not appear to be biomagnified between trophic levels.

#### 11.5 REFERENCES CITED FOR MANGANESE

Barbour, M.G., J.H. Burk, and W.D. Pitts. 1987. Terrestrial Plant Ecology, 2nd edition. Benjamin/Cummings Publishing Company, Inc. Menlo Park, Calif. 634 pp.

Bordeau, P and M. Treshow. 1978. "Ecosystem responses to pollution." In: Principles of Ecotoxicology. (G.C. Butler, ed.). SCOPE 12. J. Wiley & Sons. New York.

Davies, P.H. 1980. Water Pollution Studies: Job 2. "Investigations on the toxicity of metals to fish." Job Progress Report F-33-R-15. Colorado Division of Wildlife. March, 1980. Ft. Collins, Colo.

Donahue, R.L., R.W. Miller, and J.C. Shickluna. 1983. Soils: an introduction to soils and plant growth. Prentice-Hall, Inc. Englewood Cliffs, N.J. 667 pp.

EPA (U.S. Environmental Protection Agency). 1976.

Goyer, R.A. 1986. "Toxic effects of metals." In: Toxicology: the basic science of poisons. (C.D. Klaassen, M.O. Amdur, and J. Doull, eds.), pp.582-635. Macmillan Publishing Company. New York. 974 pp.

Hartenstein, R., A.L. Leaf, E.F. Neuhauser, and D.H. Bickelhaupt. 1980. "Composition of the earthworm *Eisenia foetida* (Savigny) and assimilation of 15 elements from sludge during growth." Comp. Biochem. Physio. 66C:187-192.

Hartenstein, R., E.F. Neuhauser, and A. Narahara. 1981. "Effects of heavy metal and other elemental additives to activated sludge on growth of *Eisenia foetida*." J. Environ. Qual. 10:372-376.

Hue, N.V. 1988. "A possible mechanism for manganese phytotoxicity in Hawaii soils amended with a low-manganese sewage sludge." J. Environ. Qual. 17(3): 473-479.

Ireland, M.P. 1979. "Distribution of essential and toxic metals in the terrestrial gastropod *Arion ater*." Environ. Pollut. 13:271-278.

Jernelov, A., K. Beijer, and L. Soderlund. 1978. "General aspects of toxicology." In: Principles of Ecotoxicology. (G.C. Butler, ed.). SCOPE 12. J. Wiley & Sons. New York.

Lewis, M.A., D.W. Evans, and J.G. Wiener. 1979. "Manganese." Pages 137-144. In: Thurston, R.V., R.C. Russo, C.M. Fetterolf, Jr., T.A. Edsall, and Y.M. Barber, Jr. (eds.). A Review of the EPA Red Book: Quality Criteria for Water. American Fisheries Society. Bethesda, Md. April 1979.

Martin, M.H. and P.J. Coughtrey. 1982. Biological monitoring of heavy metal pollution. Applied Science Publishers. London. 475 pp.

Nagao, M.A. and H.H. Hirae. 1992. "Macadamia: cultivation and physiology." Critical Reviews in Plant Sciences 10(5):441-470.

National Academy of Sciences (NAS). 1973. Manganese. NAS. Washington, D.C. 108 pp.

Salisbury, F.B. and C.W. Ross. 1985. Plant Physiology, 3rd edition. Wadsworth Publishing Company. Belmont, Calif. 540 pp.

Taylor, G.J., K.J. Stadt, and M.R.T. Dale. 1991. "Modelling the phytotoxicity of aluminum, cadmium, copper, manganese, nickel, and zinc using the Weibull frequency distribution." Canadian Journal of Botany 69(2):359-367.

## 12.0 MERCURY

Mercury (Hg) compounds have no known role in normal physiology, and their presence in the cells of living organisms apparently represents contamination from natural and anthropogenic sources. Researchers have had difficulty specifying threshold levels or toxic effects on the basis of present knowledge (NAS 1978).

### 12.1 TERRESTRIAL AND AQUATIC VEGETATION

Rai et al. (1990) found that 0.8 µg/ml Hg inhibited CO<sub>2</sub> uptake in algae, thus decreasing total algal numbers in solution. Filamentous algae were more tolerant of high Hg concentrations than were unicellular forms. Inhibition of CO<sub>2</sub> was maximum at a toxic exposure rate of 79 percent. Growth inhibition was determined for eelgrass in a seawater solution at 5 µM Hg (Lyngby et al. 1984).

Mercuric ion concentrations in water cause a considerable reduction in the photosynthetic capability of phytoplankton. Just 0.05 ppm Hg ions in water caused a 50 percent reduction in photosynthesis in *Macrocyctis pyrifers*. A concentration of 500 ppb caused a 15 percent decrease in photosynthesis in one day and complete inactivation in four days (Mitra 1986). He reported that 0.6 ppb of ethylmercury phosphate was the threshold concentration for inhibition of the growth of marine phytoplankton, and that 60 ppb was lethal to all marine species. For Hg salts, Mitra (1986) found that the threshold lethal concentration for algae ranged from 900 to 60,000 ppb. Harriss et al. (1970) subjected the marine diatom *Nitzschia delicatissima* to various concentrations of alkylmercurial fungicides. As little as 0.1 µg/l of the fungicide decreased the growth and photosynthesis of the marine diatom as well as some freshwater phytoplankton.

All plants appear to accumulate traces of Hg, but the amount depends on the plant species, locality, and chemical form of Hg available. Rooted plants absorb elemental Hg and alkylmercurials much more readily than ionic inorganic mercury (Dolar et al. 1971). Algae are especially at risk because they constitute the majority of the aquatic primary producers in the

food chain for these systems. From this initial point of entry, Hg is concentrated up the food chain.

## 12.2 TERRESTRIAL INVERTEBRATES

Although the concentration of Hg in aquatic food chains has long been recognized, its effects on terrestrial invertebrates have been studied much less. The toxic effects of Hg in animals are related to its effects on membrane function, and toxicity is therefore potentially widespread through all phyla. Because of the ability of various biota to methylate elemental Hg into more toxic and bioavailable forms, the presence of elemental Hg in soils has serious implications for associated ecosystems.

In earthworms, which are a major component of soil ecosystems and terrestrial food chains, toxic effects have been widely noted, although a few studies reported ambiguous results. In *Octochaetus pattoni* bioassayed in soil contaminated with 0 - 5 ppm Hg, the LD<sub>50</sub> was 2.39 ppm after 10 days and 0.79 after 60 days (Abbasi and Soni 1983). After 60 days, 35 percent of the animals at the 0.5 ppm level were dead, and 100 percent mortality occurred in the 5 ppm group. One study on sewage sludge reported that the Hg was not bioavailable to earthworms, based on an apparent lack of bioconcentration (Helmke et al. 1979). A different study found decreased growth rates at 480 - 4,800 mg/kg; mortality occurred at all Hg concentrations greater than 2,400 mg/kg (Hartenstein et al. 1981).

The 48-hour LC<sub>50</sub> for Hg in mosquito larvae was 0.29 ppm; no safe concentration was found. A second bioassay on *Aedes aegypti* larvae found 80 percent mortality after 24 hours and 100 percent after 72 hours in the 0.5 ppm group. At 5 ppm, mortality was 90 percent at 48 hours and 100 percent after 72 hours (Abbasi et al. 1985).

## 12.3 TERRESTRIAL VERTEBRATES

The long-term use and subsequent ban in 1966 of alkyl mercury seed dressings in Sweden has provided some valuable comparisons of Hg concentrations. Concentrations in liver, muscle, and



kidney of goshawks (*Accipiter gentilis*) in 1966 were 2.27, 0.99, and 3.06 µg/g, respectively. These same measurements, taken eight years after the ban in 1974, were 0.5, 0.2, and 0.57 µg/g (Henrikson and Karppanen 1975). In starlings, Hg concentrations in whole bodies (less bill, skin, and wings) were 0.063 µg/g at the initiation of a ban on mercurial fungicides. Two years later, values had dropped to 0.02 µg/g (White et al. 1977). Concentrations of Hg in rodent livers were 1.248 µg/g in fields treated with Hg seed dressings and 0.18 µg/g in untreated areas (Fimreite et al. 1970).

Chronic exposure of laboratory rats to inorganic Hg has resulted in decreased body weight and increased kidney weight. The central nervous system is a major target for organic Hg compounds. Adverse effects in humans to subchronic and chronic oral exposures include brain lesions, brain cell destruction, hearing and visual impairment, and loss of sensation to extremities.

#### 12.4 AQUATIC FAUNA

Measurements of Hg levels in water and sediments, though useful, are not sufficient to ascertain the rates of methylation or uptake by biota. Methylation rates in ecosystems are a function of the Hg burden, bacterial population, nutrient loadings, pH and redox condition, suspended sediment load, sedimentation rates, and other physiochemical conditions (NAS 1978). The bioaccumulation of methylmercury into the tissues of higher organisms is apparently diffusion-controlled. The diffusion rate of methylmercury chloride through cell membranes into cells is reportedly so rapid that even low concentrations of methylmercury in water can lead to elevated concentrations in fish (Rakow et al. 1977). After methylmercury diffuses through the cell membrane, it is rapidly bound by sulfhydryl groups, thereby maintaining the concentration gradient across the membrane. This means that extremely low concentrations of methylmercury will bioaccumulate rapidly in ecosystems.

Depending on the level of pollution, many invertebrate organisms tolerate and magnify various mercurials to some degree. In unpolluted Swedish waters, the level of mercury in caddisflies (Trichoptera), stoneflies (Plecoptera), alderflies (Neuroptera), and aquatic sowbugs (Isopoda:

*Asellus*) ranged between 0.025 and 0.072 µg/g. In water polluted with phenylmercuric compounds, the levels were approximately 100 times greater and ranged from 1.9 to 17 µg/g (Johnels et al. 1967, 1968; cited in NAS 1978). Although Hannerz (1968, cited in NAS 1978) found no direct correlation between an invertebrate's mercury burden and its trophic level, he did observe that predaceous insect larvae, such as dragonflies (Odonata) and alderflies (*Sialis*), accumulated more mercury than organisms which feed on decaying plants or detritus. The extent to which organisms concentrated Hg from water varied from <100 to >12,000-fold, depending on such factors as the form of the mercury, time of exposure, nature of the food consumed, feeding habits, and metabolic rates.

In *Daphnia*, methyl mercury chloride accumulates more readily than mercuric chloride. Incorporation into the food chain is facilitated at temperatures of 18° C compared to 10° C (Boudou and Ribeyre 1981). Pelagic fish caught in remote oceanic regions with up to 120 ppb Hg in muscles are evidence of the ability of Hg to bioconcentrate.

## 12.5 REFERENCES CITED FOR MERCURY

- Abbasi, S.A., P.C. Nipanay, and R. Soni. 1985. "Environmental consequences of the inhibition in hatching of pupae of *Aedes aegypti* by mercury, zinc and chromium - The abnormal toxicity of zinc." Intern. J. Environmental Studies 24:107-114.
- Abbasi, S.A., P.C. Nipanay, and R. Soni. 1988. "Studies on environmental management of mercury (II), chromium (VI) and zinc (II) with respect to the impact on some arthropods and protozoans - Toxicity of zinc." Intern. J. Environmental Studies 32:181-187.
- Abbasi, S.A. and R. Soni. 1983. "Stress-induced enhancement of reproduction in earthworm *Octochaetus pattoni* exposed to chromium (VI) and mercury (II) - Implications in environmental management." Intern. J. Environmental Studies 22:43-47.
- Boudou, A.C., and F. Ribeyre. 1981. "Comparative study of trophic transfer of 2 Hg compounds--HgCl<sub>2</sub> and CH<sub>3</sub>HgCl--between *Chlorella vulgaris* and *Daphnia magna*." Bull. Environ. Contam. Toxicol. 27:624-629.
- Dolar, S.G., D.R. Keeney, and G. Chesters. 1971. "Mercury accumulation by *Myriophyllum spicatum* L." Environmental Letters 1:191-198.

Fimreite, N., Fyfe, R.W., and J.A. Keith. 1970. "Mercury contamination of Canadian prairie seed eaters and their avian predators." Can. Field Nat. 84:269-276.

Harriss, R.C., D.B. White, and R. B. MacFarlane. 1970. "Mercury compounds reduce photosynthesis by plankton." Science 170:736-737.

Hartenstein, R., E.F. Neuhauser, and A. Narahara. 1981. "Effects of heavy metal and other elemental additives to activated sludge on growth of *Eisenia foetida*." J. Environ. Qual. 10:372-376.

Helmke, P.A., W.P. Robarge, R.L. Korotov, and P.J. Schomberg. 1979. "Effects of soil-applied sewage sludges on concentrations of elements in earthworms." J. Environ. Qual. 8:322-327.

Henrikson, K., and E. Karppanen. 1975. "Mercury residues in goshawk (*Accipiter gentilis*) in 1966 and 1974." Nord. Vet. Med. 27:641-645.

Jernelov, K. Beijer, and L. Soderlund. 1978. "General aspects of toxicology." In: Principles of Ecotoxicology (G.C. Butler, ed.). SCOPE 12. J. Wiley & Sons. New York.

Lyngby et al. 1984.

Mitra, S. 1986. Mercury in the ecosystem. Switzerland: Trans Tech Publications Ltd. 203 pp.

National Academy of Sciences (NAS). 1978. An assessment of mercury in the environment. NAS. Washington, D.C. 185 pp.

Rai, L.C., A.K. Singh, and N. Mallick. 1990. "Employment of CEPEX enclosures for monitoring toxicity of Hg and Zn on in situ structural and functional characteristics of algal communities of river Ganga in Varanasi, India." Ecotoxicology and Environmental Safety 20: 211-221.

Rakow, S. and J.R. Lakowicz. 1977. Membrane permeability to methylmercuric chloride by fluorescence quenching. University of Minnesota, Freshwater Biological Institute.

White, D.H., J.R. Bean, and J.R. Longcore. 1977. "Nationwide residues of mercury, lead, cadmium, arsenic, and selenium in starlings, 1973." Pestic. Monitor J. 11:35-39.

### 13.0 SILVER

Silver (Ag) is very toxic to living organisms in minute amounts. Silver is a white, ductile metal occurring naturally in the pure form and in ores. Bowen (1979) suggested that Ag shows a close

relationship between parent material and soil concentrations. This may have some relationship to bioaccumulation over time. Reported concentrations of Ag in parent materials are on the order of 0.05 µg/g, slightly lower than the average crustal abundance. Near smelters, power plants, and in sewage sludges, values of 0.3 µg/g can be expected. Although highly toxic to plants, there appear to be no relevant data concerning its chemistry or mobility in soils (Coughtrey and Thorne 1983).

### 13.1 TERRESTRIAL AND AQUATIC VEGETATION

The silver ion  $\text{Ag}^+$  is an effective inhibitor of ethylene action in plants (Beyer 1976). Among the ethylene effects found by Beyer to be nullified or inhibited by the Ag ion were the etiolation of pea seedlings; promotion of abscission of leaves, flowers, and fruits of cotton; and induction of senescence in orchid flowers. Silver thiosulfate has proven to be even more effective in delaying senescence of cut flowers than Ag nitrate (Halevy and Mayak 1981).

Stokes (1973) found that 30 µg/l of Ag inhibited the growth of the alga *Chlorella vulgaris* in solution. The EPA (1980) found that toxicity of Ag to 13 freshwater plant species occurred at 30 to 7,500 µg/l. It appears that the adverse effects of Ag on plants are unlikely at and below concentrations not harmful to freshwater animals. Therefore, plants are probably more resistant to Ag than some animals, and thus their well-being is assured if the more sensitive animals are protected.

Hunter (1953) studied seasonal changes in the concentrations of many elements, including Ag, in fronds and rhizomes of the fern *Pteridium aquilinum*. He noted that concentrations of Ag increased gradually and were highest when the fronds were old, presumably because of exposure time. Bioaccumulation of Ag in plants apparently does occur. However, little work has been done on its movement through the food chain.

## 13.2 TERRESTRIAL INVERTEBRATES

The effect of Ag on terrestrial invertebrates has not been well studied, and no literature was found on this topic.

## 13.3 TERRESTRIAL VERTEBRATES

Silver does not occur regularly in animal tissues. The major effect of excessive absorption of Ag is local or generalized impregnation of the tissues, where it remains as Ag sulfide. This forms an insoluble complex in elastic fibers, resulting in argyria (Goyer 1986). Although the data for the systemic distribution of stable Ag are variable, they do not suggest that any organ or tissue, except perhaps the spleen, concentrates the element to any great extent (Coughtrey and Thorne 1983).

## 13.4 AQUATIC FAUNA

Silver exhibits oxidation states of 0, +1, +2, and +3, but only the 0 and +1 states occur to any extent in the environment. In natural water, the monovalent species is the form of environmental concern. Monovalent Ag ions may exist in various degrees of association with a large number of inorganic ions, such as sulfate, bicarbonate, and nitrate, to form numerous compounds with a range of solubilities and potentials for hydrolysis or other reactions (EPA 1980). Most of the toxicity studies have been conducted with Ag nitrate, which is an excellent source of free soluble Ag ions.

The data concerning acute toxicity of Ag to freshwater organisms include 82 values for 10 species from nine different taxonomic families (EPA 1980). Water hardness and chloride concentration are the two factors involved with acute Ag toxicity in aquatic organisms. For invertebrate species, acute values for Ag range from 0.25 µg/l for the water flea *Daphnia magna* to 4,500 µg/l for the scud *Gammarus pseudolimnaeus* (EPA 1980). Acute values for fish range from 3.9 µg/l for the fathead minnow in soft water to 280 µg/l for rainbow trout in hard water. It appears that Ag is more toxic in soft water.

The available data indicate that acute toxicity to freshwater aquatic life may occur at concentrations of 1.2 µg/l in solution (water hardness of 50), and chronic toxicity at concentrations as low as 0.12 µg/l (EPA 1980). Chronic values as high as 29 µg/l were determined in the laboratory. No information was found concerning the relationship between water hardness and chronic Ag toxicity.

Silver seems to bioaccumulate to some degree in food chains. The bioconcentration factors for Ag range from less than one for bluegills to 240 for insect larvae (EPA 1980). Little information for bioaccumulation of Ag in food web matrixes exists. Limited information is available concerning the relationship of various forms of Ag and toxicity to aquatic animals.

### 13.5 REFERENCES CITED FOR SILVER

Beyer, E.M., Jr. 1976. "A potent inhibitor of ethylene action in plants." Plant Physiology 58:268-271.

Bowen, H.J.M. 1979. Environmental Chemistry of the Elements. Academic Press. London.

Coughtrey, P.J., and M.C. Thorne. 1983. Radionuclide distribution and transport in terrestrial and aquatic ecosystems: a critical review of data. Vol. 2. A.A. Balkema. Rotterdam, Netherlands. 2,500 pp.

EPA (U.S. Environmental Protection Agency). 1980. "Ambient Water Quality Criteria for Silver." EPA PB81-117822. U.S. Environ. Prot. Agency, Washington, D.C. Halevy, A.H. and S. Mayak. 1981. "Senescence and postharvest physiology of cut flowers--Part 2." Horticultural Reviews 3:59-143.

Goyer, R.A. 1986. "Toxic effects of metals." In: Toxicology: the basic science of poisons. (C.D. Klaassen, M.O. Amdur, and J. Doull, eds.), pp.582-635. Macmillan Publishing Company. New York. 974 pp.

Hunter, J.G. 1953. "The composition of bracken: Some major and trace element constituents." J. Sci. Food Agric. 4:10-20.

Stokes, P.M. 1973. "Heavy-metal tolerance in algae isolated from contaminated lakes near Sudbury, Ontario." Can. Jour. Bot. 51:2155.

## 14.0 ZINC

Zinc (Zn) is an essential metal, acting as a cofactor in many enzymes. Thus, it is not surprising that many organisms have systems to accumulate and store Zn. However, at concentrations above the micronutrient level required, Zn exerts toxic effects. Zinc may enter a food chain through aerial deposition on foliage or through uptake by plant roots. Although Zn is extremely soluble, uptake by roots is limited.

### 14.1 TERRESTRIAL AND AQUATIC VEGETATION

Zinc is an important micronutrient for plants. It is essential to the synthesis of the important plant hormone indole acetic acid (IAA) and may be involved in protein synthesis (Barbour et al. 1987).

Fungal hyphae of mycorrhizae growing from the plant roots into additional soil areas help to absorb many nutrients, particularly the less mobile nutrients such as Zn (Donahue et al. 1983). Excess soil phosphorus can cause Zn deficiency. In susceptible plants, such as corn, beans, and flax, excess soluble phosphate precipitates Zn into insoluble Zn phosphates, both inside the plant and in the soil (Donahue et al. 1983). As with most metals, Zn is interactive with other elements in the soil. Micronutrient cations such as Zn, are relatively insoluble in nutrient solutions when provided as common inorganic salts, and they are nearly insoluble in most soil solutions (Salisbury et al. 1985). This insolubility is especially marked if the pH is above 5 (Clark 1982, Vose 1982).

Taylor et al. (1991) subjected wheat to various Zn concentrations in soil media. The wheat plants showed signs of growth reduction at a threshold of 37  $\mu\text{M}$ . Using the Weibull frequency distribution, Zn caused 0.5 percent growth reduction/ $\mu\text{M}$ . Growth reduction occurred even at nearly neutral pH conditions (6.5) in the soil.

Surface application of Zn on rangelands having claypan soils could increase herbage production, but the Zn concentration could become toxic to the crown and roots of the grasses. White (1991)

found that herbage decreased and chlorosis occurred in blue grama (*Bouteloua gracilis*) plants when application rates exceeded 0.40 g Zn/kg soil. At 2.0 g Zn/kg soil applied as ZnCl<sub>2</sub>, one-half of the plants died (White 1991).

Rai et al. (1990) found that a concentration of 0.8 µg/ml in solution maximized inhibition of algal numbers. Inhibition of CO<sub>2</sub> uptake by phytoplankton in Ganges River water was maximum at 0.8 µg/ml Zn (69 percent). In a similar study, Lyngby and Brix (1984) found that eelgrass in seawater solution had inhibited growth at a concentration of 50 µM. Bioaccumulation and toxicity of Zn in *Cladophora glomerata* from two populations in the River Roding, United Kingdom, were examined in experimental laboratory flowing-water channels. Little difference was found in Zn bioaccumulation between *Cladophora* from the site showing mild organic pollution and the site subjected to considerable inputs from urban and motorway runoff. Uptake of Zn increased with increasing concentration in the test solution and was linear and proportional up to 0.4 mg/l. Three stages of uptake were identified, with the most dramatic accumulation occurring in the first 10 minutes (McHardy and George 1990). Some bioaccumulation work has been done with Zn, but most emphasis has been placed on Zn deficiencies in agronomic plants. It appears that accumulation of Zn increases at a faster rate than toxicity as plants are exposed to increasing environmental concentrations. Thus, bioaccumulation up the food chain may be of concern (Kelly and Whitton 1989).

#### 14.2 TERRESTRIAL INVERTEBRATES

The great majority of studies report only Zn concentrations in tissue and the potential for bioconcentration above background levels. Most authors report a correlation between soil levels and tissue levels of the metal; however, this relationship is not supported in all studies. Certainly the feeding strategy, assimilation efficiency, and physiology of the animal may influence the endogenous level of Zn.

Zinc transfer through a food chain was observed along a gradient from an old mine site (Roberts and Johnson 1978). Zinc levels were tabulated in a variety of herbivorous invertebrates,



including grasshoppers, beetles, earwigs, springtails, bugs, flies, and ants. Zinc concentration in these primary consumers was similar to that found in cover vegetation (220 µg/g; control levels approximately 55 µg/g). Zinc concentration increased in such secondary consumers as spiders, beetles, and centipedes (280 µg/g; controls at 50 µg/g).

In five species of phytophagous insects (three Hymenoptera and two Lepidoptera), Zn concentrations were much greater than any other metal studied, and 100 times higher than Cd concentrations (Lindqvist 1992). In all cases, body levels of Zn were higher in larvae than in the food plants they consumed and higher in adults than in larvae.

Earthworms of three genera (*Lumbricus*, *Alabophora*, and *Octoclasium*) from six undisturbed soil types showed concentration ratios of 3 to 13 (Van Hook 1974). Typically, concentration ratios are higher in worms from less polluted soils, reinforcing the idea that many organisms actively control accumulation of this metal. In *Lumbricus terrestris*, tissue concentrations correlated with distance from an old mine site (highest Zn concentration =  $210.0 \pm 37.2$  µg/g; control =  $90.0 \pm 7.02$  µg/g). *Lumbricus rubellus* taken directly from the soil at an abandoned mine showed much higher Zn concentrations:  $2,511.4 \pm 305.6$  µg/g ( $426.1 \pm 34.2$  µg/g at control site). When localization studies were done, the greatest concentration of Zn was observed in the posterior alimentary canal (Morgan and Morgan 1990).

Worms of three genera (*Lumbricus*, *Dendrobaena*, and *Octoclasium*) from a smelter site showed a strong correlation between body weight and Zn concentration; the highest correlation was seen closest to the smelter (C.H. Jones, unpub., reported in Martin and Coughtrey 1982). Population densities of nine worm species from two genera (*Lumbricus* and *Allolobophora*) were lower at a smelter site than at a control site (64 worms/m<sup>2</sup> versus 161.8 worms/m<sup>2</sup>). Zinc concentrations in worms ranged from 634 - 1,398 at the smelter and 264 - 914 at the control site; concentration factors ranged from 1.03 - 2.26 at the smelter and 2.97 - 10.25 at the control site (Wright and Stringer 1980).

*Dendrobaena rubida* from a mine site showed Zn levels of  $584 \pm 110$  ppm; Zn was concentrated 58 times above the level found in the mine site soil compared to a six-fold elevation in worms from the control site (Ireland 1975). *D. rubida* taken from contaminated soil and placed on control soil showed control levels of Zn after five days. Interestingly, control worms moved to contaminated soil showed no increase in Zn concentrations. Worms reared in sewage sludge and observed for uptake and loss of Zn showed that the metal accumulated in a linear fashion; loss followed first order kinetics. After approximately 30 days, half of the Zn had been eliminated (Helmke et al. 1979).

In two studies of worms used to digest sewage sludge (*Eisenia foetida*), Hartenstein et al. (1980, 1981) reported increased Zn concentration after exposure to sludge containing Zn; the concentration factors decreased with increasing applications of Zn over a 5-week period. Concentration ratios were 0.06 at 2,500 ppm of Zn acetate applied, 0.03 at 5,000 ppm, and 0.02 at 10,000 ppm. Zinc at concentrations of 1,300-13,000 mg/kg inhibited growth of earthworms. Mortality was seen at Zn concentrations greater than 26,000 but was not definitively associated with the metal (Martin and Coughtrey 1982).

As herbivores and detritivores, mollusks play an important role in most terrestrial ecosystems. Their niche also allows them to introduce heavy metals into terrestrial food chains. Many studies focused on either snails or slugs found similar patterns in the two groups. These studies report content and tissue concentration for Zn, and some have reported small bioconcentration factors for Zn. Typically, concentration ratios are lower at sites with higher levels of Zn.

Studies of slugs from three genera (*Deroceras*, *Arion*, and *Milax*) at old mine sites revealed Zn concentrations of  $586.5 \pm 69.1$  to  $1257.7 \pm 152.6$   $\mu\text{g/g}$  (Greville and Morgan 1989, 1990). These levels in tissue reflect concentration ratios of 1.85 above soil or vegetation and 0.48 from litter. *Arion ater* at an old mine site stored most Zn in digestive glands; total body burden was  $1.23 \text{ mg/g} \pm 0.06$ ; background levels were  $0.28 \pm 0.02$  (Ireland 1979).

Concentration ratios for the snail *Helix aspersa* were 2.94 above soil or vegetation and 0.09 above litter at a smelter site. The maximum concentration reported for the snail was  $403 \mu\text{g/g} \pm 150.5$  (Coughtrey and Martin 1976). This sample was quite variable, with a coefficient of variation of 37 percent, although there was a significant correlation between Zn concentration and body weight ( $r = 0.954$ ). This correlation between body weight and concentration was also found in *Cepaea hortensis*, in which it was found that variance in body weight accounted for 67 percent of the variance in Zn concentration (Williamson 1980). Tissue Zn concentration in these snails dropped slightly after collection from a contaminated site and subsequently leveled off without progressive elimination. As with *Arion ater* (Ireland 1979), the bulk of the total body burden was found in the digestive gland.

Concentration factors for snails of three genera (*Clausilia*, *Helix*, and *Cepaea*) ranged from 0.52 - 1.15 for uncontaminated sites to 0.09 - 0.56 for moderately contaminated sites to 0.80 - 1.28 for the most contaminated sites at an old Zn mine and a smelter (Coughtrey and Martin 1975, 1976). Slugs from two genera (*Arion* and *Agriolimax*) showed a similar pattern: Concentration ratios from the least contaminated sites were 1.64 - 4.83; from moderately contaminated sites were 0.14 - 2.62; and from most contaminated sites were 0.48 - 1.85. The concentration ratios were also lower for a given species when the animal was allowed to forage on litter as opposed to living vegetation (Coughtrey and Martin 1982).

Many arthropods, as with mollusks and earthworms, are important recyclers of nutrients and thus may be expected to accumulate heavy metals from their environment. Bioassays of two species, *Folsomia fimetarioides* and *Isotomiella minor*, revealed that tissue burdens of Zn were greatest when the animals were reared on contaminated soil and fungi ( $20,740 \mu\text{g/g}$ ) (Tranvik and Eijsackers 1989). When contaminated fungi were fed to these collembolans, Zn concentrations were  $2,000 \mu\text{g/g}$  (controls had  $1,760 \mu\text{g/g}$ ). Survivorship for each species declined under drought stress conditions regardless of the environmental contaminant load.

Woodlice *Oniscus asellus* are important detritus feeders in many ecosystems. Like snails and slugs, they typically bioconcentrate Zn above levels in soil or on vegetation near contaminated sites (BCFs of 1.37 - 1.63), although in litter the ratios were less than one (0.05 - 0.30) (Martin and Coughtrey 1976). In one instance, the concentration of Zn in decaying litter near a Zn smelter was very similar to that found in the isopods. The mean Zn concentration from the isopods *Porcellio scaber* and *O. asellus* from a variety of sites was  $1.47 \pm 0.09$  µg/g (range 0.17 - 7.41 µg/g). The mean concentration ratio for these animals was  $0.75 \pm 0.06$  (range 0.08-3.03) (Martin and Coughtrey 1982). A strong correlation was seen between Zn concentration and body weight,  $r=0.76$ .

Like other organisms, isopods probably actively regulate endogenous Zn levels (Coughtrey et al. 1980). Unlike many arthropods, isopods do not store Zn in their exoskeleton. The hepatopancreas is the primary storage organ for Zn in isopods, where up to 76 percent of the Zn may be held. In isopods taken from many sites, including old mines, the maximum Zn concentration was found to be  $178 \pm 14$  ppm (54.3 at control sites); no toxic effects were seen in any of these animals (Hopkin and Martin 1982).

Mosquito (*Aedes aegypti*) larvae treated in experimental flasks containing 0 - 50 ppm Zn experienced no mortality after 48 hours (Abbasi et al. 1988). Pupae placed in flasks containing Zn solutions exhibited 20 percent mortality after 48 hours at 0.5 ppm and 30 percent mortality after 48 hours at 5 ppm (Abbasi and Soni 1983). Despite the relatively high survivorship in both treatment groups, behavioral and/or physiological abnormalities that prevented them from swimming or flying as adults would have resulted in much higher mortality under natural conditions.

### 14.3 TERRESTRIAL VERTEBRATES

Beyer et al. (1985) found that very little of the Zn in soil was incorporated in flora and fauna; contamination came predominantly from aerial deposition. They also found higher concentrations of Zn in shrews and lower concentrations in mice, in contrast to Roberts and Johnson (1978),

who found similar values between these herbivores and insectivores. Kidney concentrations in gray squirrels were higher in urban areas (25.5 to 31.9 µg/g) than in rural areas (14.3 to 18.6 µg/g) (McKinnon et al. 1976).

Zinc absorption is affected by numerous dietary factors. These interactions, and the uptake mechanisms, are generally not well understood. In a laboratory study, Zn was administered in drinking water (200 mg/l) by itself and in combination with other metals (Cooke et al. 1990). Resultant Zn concentrations in the kidneys were higher than liver and femur concentrations. However, this was also the case when the combinations zinc/cadmium and iron/lead/zinc/cadmium were administered. In fact, the highest kidney concentrations occurred in the high Cd-only treatments. This may reflect the induction of metallothioneins, which can bind Zn and Cd, and subsequent redistribution and accumulation in the kidney (Cooke et al. 1990).

Zinc seems to have a very low level of transfer potential through terrestrial food chains, which may be associated with its essential role in biological systems (Roberts and Johnson 1978).

#### 14.4 AQUATIC FAUNA

Zinc is an essential trace element for animals and is important to cell growth and differentiation and the formation of a number of metalloenzymes (NAS 1980, Rand and Petrocelli 1985). In aquatic systems, acute toxicity to fish includes gill destruction and hypoxia (Rand and Petrocelli 1985). Exposure of fish to sublethal concentrations of Zn can cause extensive edema and necrosis of liver tissue (Rand and Petrocelli 1985). Water quality criteria have been developed by EPA (1986) based on the relationship of decreasing Zn toxicity with increasing water hardness.

Cladocerans are the most sensitive aquatic animal species to Zn (EPA 1987). The mean acute value determined, at a water hardness of 50 mg/l, was 93.95 µg/l. *Daphnia* sp. had a mean acute toxicity value of 299.8 µg/l. *Argia* sp. were the most tolerant animal, with an acute value of

88,960 µg/l (EPA 1987). Trout are among the most sensitive fish tested in acute assays with Zn (EPA 1987). The mean acute value, at water hardness of 50 mg/l, for rainbow trout is 689.3 µg/l (EPA 1987). Davies (1980) has reported a 96-hour LC<sub>50</sub> for rainbow trout (170 mm) of 105 µg/l (water hardness was 36.7 mg/l) and 186 µg/l (water hardness was 39.2 mg/l) in aerated and nonaerated tests, respectively.

A concentration of 47 µg/l was the lowest maximum acceptable toxicant concentration (MATC) for *Daphnia magna* at a water hardness of 104 mg/l (EPA 1986). The flagfish (*Jordanella floridae*) had an MATC of 36.4 µg/l (water hardness of 44 mg/l) and was the most sensitive of seven fish species tested (EPA 1986). Trout are apparently not as sensitive to the chronic effects of Zn as the flagfish. The chronic value for brook trout was 854.7 µg/l at water hardness of 45.9 mg/l (Holcombe et al. 1979, cited in EPA 1987). Chronic values based on early life cycle tests with rainbow trout are 276.7 µg/l (water hardness = 26 mg/l) (Sinley et al. 1974, cited in EPA 1987) and 603.0 µg/l (hardness = 25 mg/l) (Cairns et al. 1982, cited in EPA 1987).

Zinc has shown bioconcentration factors of 51 to 1,000 in freshwater fish (EPA 1986, 1987), although limited information is available. A whole body bioconcentration factor of 417.3 was reported for *Jordanella floridae* following 100 days of exposure (Spehar 1978, cited in EPA 1987).

#### 14.5 REFERENCES CITED FOR ZINC

Abbasi, S.A., P.C. Nipanay, and R. Soni. 1988. "Studies on environmental management of mercury (II), chromium (VI), and zinc (II) with respect to the impact on some arthropods and protozoans - Toxicity of zinc." Intern. J. Environmental Studies 32:181-187.

Abbasi, S.A. and R. Soni. 1983. "Stress-induced enhancement of reproduction in earthworm *Octochaetus pattoni* exposed to chromium (VI) and mercury (II) - Implications in environmental management." Intern. J. Environmental Studies 22:43-47.

Barbour, M.G., J.H. Burk, and W.D. Pitts. 1987. Terrestrial Plant Ecology, 2nd edition. Benjamin/Cummings Publishing Company, Inc. Menlo Park, Calif. 634 pp.

Beyer, W.N., O.H. Pattee, L. Sileo, D.J. Hoffman, and B.M. Mulhern. 1985. "Metal contamination in wildlife living near two zinc smelters." Environ. Pollut. 38 A:63-86.

Bordeau, P. and M. Treshow. 1978. "Ecosystem responses to pollution." In: Principles of Ecotoxicology (G.C. Butler, ed.). SCOPE 12. J. Wiley & Sons. New York.

Clark, R.B. 1982. "Iron deficiency in plants grown in the Great Plains of the U.S." Journal of Plant Nutrition 5:251-268.

Cooke, J.A., S.M. Andrews, and M.S. Johnson. 1990. "The accumulation of lead, zinc, cadmium and fluoride in the wood mouse (*Apodemus sylvaticus* L.)." Water, Air and Soil Pollution 51:55-63.

Coughtrey, P.J. and Martin, M.H. 1976. "The distribution of lead, zinc, cadmium and copper within the pulmonate mollusc, *Helix aspersa* Muller." Oecologia 23:315-322.

Coughtrey, P.J. and Martin, M.H. 1977. "The uptake of lead, zinc, cadmium, and copper by the pulmonate mollusc, *Helix aspersa* Muller, and its relevance to the monitoring of heavy metal contamination of the environment." Oecologia 27:65-74.

Coughtrey, P.J., M.H. Martin, J. Chard, and S.W. Shales. 1980. "Microorganisms and metal retention in the woodlouse *Oniscus asellus*." Soil Biol. Biochem. 12:23-27.

Davies, P.H. 1980. Water Pollution Studies: Job 2. "Investigations on the toxicity of metals to fish." Job Progress Report F-33-R-15. Colorado Division of Wildlife. March, 1980. Ft. Collins, Colo.

Donahue, R.L., R.W. Miller, and J.C. Shickluna. 1983. Soils: an introduction to soils and plant growth, fifth edition. Prentice-Hall, Inc. Englewood Cliffs, N.J. 667 pp.

Greville, R.W. and A.J. Morgan. 1989. "Concentrations of metals (Cu, Pb, Cd, Zn, Ca) in six species of British terrestrial gastropods near a disused lead and zinc mine." J. Moll. Stud. 55:31-36.

Greville, R.W. and A.J. Morgan. 1990. "The influence of size on the accumulated amounts of metals (Cu, Pb, Cd, Zn and Ca) in six species of slug sampled from a contaminated woodland." Moll. Stud. 56:355-362.

Greville, R.W. and A.J. Morgan. 1991. "A comparison of (Pb, Cd and Zn) accumulation in terrestrial slugs maintained in microcosms: Evidence for metal tolerance." Env. Poll. 74:115-127.

Hartenstein, R. Neuhauser, and J. Collier. 1980. "Accumulation of heavy metals in the earthworm *Eisenia foetida*." J. Environ. Qual. 9:23-26.

Hartenstein, R., E.F. Neuhauser, and A. Narahara. 1981. "Effects of heavy metal and other elemental additives to activated sludge on growth of *Eisenia foetida*." J. Environ. Qual. 10:372-376.

Helmke, P.A., W.P. Robarge, R.L. Korotov, and P.J. Schomberg. 1979. "Effects of soil-applied sewage sludges on concentrations of elements in earthworms." J. Environ. Qual. 8:322-327.

Hopkin, S.P. and M.H. Martin. 1982. "The distribution of zinc, cadmium, lead and copper within the woodlouse *Oniscus asellus* (Crustacea, Isopoda)." Oecologia 54:227-232.

Ireland, M.P. 1975. "Distribution of lead, zinc and calcium in *Dendrobaena rubida* (Oligochaeta) living in soil contaminated by base metal mining in Wales." Comp. Biochem. Physiol. 52B:551-555.

Ireland, M.P. 1975. "Metal content of *Dendrobaena rubida* (Oligochaeta) in a base metal mining area." Oikos 26:74-79.

Ireland, M.P. 1979. "Distribution of essential and toxic metals in the terrestrial gastropod *Arion ater*." Environ. Pollut. 13:271-278.

Joose, E.N.G., K.J. Wulfraat, and H.P. Glas. 1981. "Tolerance and acclimation to zinc of the isopod *Porcellio scaber* Latr. In: International Conference." Heavy Metals in the Environment pp. 425-428. Amsterdam, September, 1981. CEP Consultants, Ltd, Edinburgh.

Kelly, M.G. and B.A. Whitton. 1988. "Relationship between accumulation and toxicity of zinc in *Stigeoclonium* (Chaetophorales, Chlorophyta)." Phycologia 28:512-517.

Lindqvist, L. 1992. "Accumulation of cadmium, copper, and zinc in five species of phytophagous insects." Env. Ent. 21:160-163.

Lyngby, J.E. and H. Brix. 1984. "The uptake of heavy metals in eelgrass *Zostera marina* and their effect on growth." Ecological Bulletins 36:81-89.

Martin, M.H. and P.J. Coughtrey. 1976. "Comparisons between the levels of lead, zinc and cadmium within a contaminated environment." Chemosphere 1:15-20.

Martin, M.H. and P.J. Coughtrey. 1982. Biological monitoring of heavy metal pollution. Applied Science Publishers. London. 475 pp.

Martin, M.H., P.J. Coughtrey, and E.W. Young. 1976. "Observations on the availability of lead, zinc, cadmium and copper in woodland litter and the uptake of lead, zinc and cadmium by the woodlouse, *Oniscus asellus*." Chemosphere 5:313-318.



McHardy, B.M. and J.J. George. 1990. "Bioaccumulation and toxicity of zinc in the green alga, *Cladophora glomerata*." Environmental Pollution 66:55-66.

McKinnin, J.G., G.L. Hoff, W.J. Bigler, and E.C. Prather. 1976. "Heavy metal concentrations in kidneys of urban grey squirrels." J. Wildlife Diseases 12:367-371.

Morgan, J.E. and A.J. Morgan. 1990. "The distribution of cadmium, copper, lead, zinc and calcium in the tissues of the earthworm *Lumbricus rubellus* sampled from one uncontaminated and four polluted soils." Oecologia 84:559-566.

National Academy of Sciences. 1980. "Mineral Tolerance of Domestic Animals. Subcommittee on mineral toxicity in animals." National Research Council. Washington, D.C.

Rai, L.C., A.K. Singh, and N. Mallick. 1990. "Employment of CEPEX enclosures for monitoring toxicity of Hg and Zn in situ structural and functional characteristics of algal communities of river Ganga in Varanasi, India." Ecotoxicology and Environmental Safety 20: 211-221.

Rand, G.M. and S.R. Petrocelli. 1985. Fundamentals of Aquatic Toxicology. Hemisphere Publishing Corp., Washington, D.C.

Roberts, R.D. and M.S. Johnson. 1978. "Dispersal of heavy metals from abandoned mine workings and their transference through terrestrial food chains." Env. Pollution 16:293-310.

Salisbury, F.B. and C.W. Ross. 1985. Plant Physiology, 3rd edition. Wadsworth Publishing Company. Belmont, CA. 540 pp.

Taylor, G.J., K.J. Stadt, and M.R.T. Dale. 1991. "Modelling the phytotoxicity of aluminum, cadmium, copper, manganese, nickel, and zinc using the Weibull frequency distribution." Canadian Journal of Botany 69(2):359-367.

Tranvik, L. and H. Eijssackers. 1989. "On the advantage of *Folsomia fimetarioides* over *Isotomiella minor* (Collembola) in a metal polluted soil." Oecologia 80:195-200.

EPA. 1986. "Quality criteria for water 1986." Office of Water Regulations and Standards. May 1, 1986. EPA 440/5-86-001. Washington, D.C.

EPA. 1987. "Ambient Water Quality Criteria for Zinc--1987." Office of Water Regulations and Standards. EPA 440-5-87-003. PB87-153581. Washington, D.C.

Van Hook, R.I. 1974. "Cadmium, lead and zinc distributions between earthworms and soils: Potentials for biological accumulation." Bull. Environ. Contam. Toxicol. 12:509-512.

Vose, P.B. 1982. "Iron nutrition in plants: a world overview." Journal of Plant Nutrition 5:233-249.

White, E.M. 1991. "Blue grama response to Zn source and rates." Journal of Range Management 44(1):48-51.

Williamson, P. 1980. "Variables affecting body burdens of lead, zinc and cadmium in a roadside population of snails *Cepaea hortensis* Muller." Oecologia 44:213-220.

Wright, M.A. and A. Stringer. 1980. "Lead, zinc and cadmium content of earthworms from pasture in the vicinity of an industrial smelting complex." Environ. Pollut. 23A:313-322.

*Appendix E*  
*Environmental Evaluation*

**ATTACHMENT E.B**

**TISSUE ANALYSES**

TISSUE ANALYSIS FROM RFP 1991\*

KEY			SUBTYPE	
SPECIES CODE	COMMON NAME	SCIENTIFIC NAME	CODE	
ACRI	GRASSHOPPER	ACRIDIDAE	AV = BIRDS	
AMTI	TIGER SALAMANDER	AMBYSTOMA TIGRINUM	BM = BENTHIC MACROINVERTEBRATES	
ARLU	WHITE SAGE	ARTEMISIA LUDOVICIANA	FI = FISH	
ASFA	ASTER	ASTER FALCATUS	HE = HERPTILES	
BOGR	BLUE GRAMA	BOUTELOUA GRACILIS	SM = SMALL MAMMALS	
BRCA	CANADA GOOSE	BRANTA CANADENSIS	TA = TERRESTRIAL ARTHROPODS	
BRIN	SMOOTH BROME	BROMUS INERMIS	VE = VEGETATION	
CACO	WHITE SUCKER	CATOSTOMUS COMMERSONI		
CRAYF	CRAYFISH			
LECY	GREEN SUNFISH	LEPOMIS CYANELLUS		
MEOF	SWEET CLOVER	MELILOTUS OFFICINALE		
MIPE	MEADOW VOLE	MICROTUS PENNSYLVANICUS		
MISA	LARGEMOUTH BASS	MICROPTERUS SALMOIDES		
NOCR	GOLDEN SHINER	NOTEMIGONUS CRYSOLEUCAS		
PEMA	DEER MOUSE	PEROMYSCUS MANICULATUS		
PIPR	FATHEAD MINNOW	PIMEPHALES PROMELAS		
POCO	CANADA BLUEGRASS	POA COMPRESSA		
SEAT	CREEK CHUB	SEMOTILUS ATROMACULATUS		
THRA	PLAINS GARTER SNAKE	THAMNOPHIS RADIX		

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MA01A	SM	BI00191EB	Cadmium	HYDRIC	2.3	MG/KG	U
MA01A	SM	BI00191EB	Chromium	HYDRIC	3.4	MG/KG	U
MA01A	SM	BI00191EB	Copper	HYDRIC	19.8	MG/KG	
MA01A	SM	BI00191EB	Lead	HYDRIC	2.9	MG/KG	
MA01A	SM	BI00191EB	Mercury	HYDRIC	0.4	MG/KG	U
MA01A	SM	BI00191EB	Selenium	HYDRIC	4.5	MG/KG	U
MA01A	SM	BI00191EB	Silver	HYDRIC	2.3	MG/KG	U
MA01A	SM	BI00191EB	Zinc	HYDRIC	90.8	MG/KG	
MA01A	VE	BI00289EB	Cadmium	HYDRIC	0.4	MG/KG	U
MA01A	VE	BI00289EB	Chromium	HYDRIC	1.2	MG/KG	
MA01A	VE	BI00289EB	Copper	HYDRIC	3.3	MG/KG	
MA01A	VE	BI00289EB	Lead	HYDRIC	0.7	MG/KG	I
MA01A	VE	BI00289EB	Mercury	HYDRIC	0.0	MG/KG	U
MA01A	VE	BI00289EB	Selenium	HYDRIC	0.8	MG/KG	U
MA01A	VE	BI00289EB	Silver	HYDRIC	0.4	MG/KG	U
MA01A	VE	BI00289EB	Zinc	HYDRIC	13.6	MG/KG	
MA01A	VE	BI00290EB	Cadmium	HYDRIC	0.4	MG/KG	U
MA01A	VE	BI00290EB	Chromium	HYDRIC	0.6	MG/KG	
MA01A	VE	BI00290EB	Copper	HYDRIC	9.9	MG/KG	
MA01A	VE	BI00290EB	Lead	HYDRIC	0.5	MG/KG	I
MA01A	VE	BI00290EB	Mercury	HYDRIC	0.1	MG/KG	U
MA01A	VE	BI00290EB	Selenium	HYDRIC	0.8	MG/KG	U
MA01A	VE	BI00290EB	Silver	HYDRIC	0.4	MG/KG	U
MA01A	VE	BI00290EB	Zinc	HYDRIC	9.3	MG/KG	
MA01A/3A	SM	BI00524EB	Cadmium	HYDRIC	2.4	MG/KG	U
MA01A/3A	SM	BI00524EB	Chromium	HYDRIC	4.9	MG/KG	
MA01A/3A	SM	BI00524EB	Copper	HYDRIC	19.4	MG/KG	
MA01A/3A	SM	BI00524EB	Lead	HYDRIC	3.8	MG/KG	
MA01A/3A	SM	BI00524EB	Mercury	HYDRIC	0.6	MG/KG	U
MA01A/3A	SM	BI00524EB	Selenium	HYDRIC	4.8	MG/KG	U
MA01A/3A	SM	BI00524EB	Silver	HYDRIC	2.4	MG/KG	U
MA01A/3A	SM	BI00524EB	Zinc	HYDRIC	110	MG/KG	
MA01R	SM	BI00247EB	Cadmium	HYDRIC	1.9	MG/KG	
MA01R	SM	BI00247EB	Chromium	HYDRIC	9.7	MG/KG	
MA01R	SM	BI00247EB	Copper	HYDRIC	26.2	MG/KG	
MA01R	SM	BI00247EB	Lead	HYDRIC	1.7	MG/KG	
MA01R	SM	BI00247EB	Mercury	HYDRIC	0.6	MG/KG	U
MA01R	SM	BI00247EB	Selenium	HYDRIC	5.9	MG/KG	U
MA01R	SM	BI00247EB	Silver	HYDRIC	4.4	MG/KG	U
MA01R	SM	BI00247EB	Zinc	HYDRIC	157	MG/KG	
MA01R	TA	BI00328EB	Cadmium	HYDRIC	3.8	MG/KG	
MA01R	TA	BI00328EB	Chromium	HYDRIC	5.4	MG/KG	U
MA01R	TA	BI00328EB	Copper	HYDRIC	51.8	MG/KG	
MA01R	TA	BI00328EB	Lead	HYDRIC	2.9	MG/KG	
MA01R	TA	BI00328EB	Mercury	HYDRIC	0.8	MG/KG	U
MA01R	TA	BI00328EB	Selenium	HYDRIC	7.1	MG/KG	U
MA01R	TA	BI00328EB	Silver	HYDRIC	3.6	MG/KG	U
MA01R	TA	BI00328EB	Zinc	HYDRIC	144	MG/KG	
MA01R	VE	BI00329EB	Cadmium	HYDRIC	0.4	MG/KG	U
MA01R	VE	BI00329EB	Chromium	HYDRIC	1.1	MG/KG	
MA01R	VE	BI00329EB	Copper	HYDRIC	3.1	MG/KG	
MA01R	VE	BI00329EB	Lead	HYDRIC	0.4	MG/KG	I
MA01R	VE	BI00329EB	Mercury	HYDRIC	0.0	MG/KG	U
MA01R	VE	BI00329EB	Selenium	HYDRIC	0.8	MG/KG	UI
MA01R	VE	BI00329EB	Silver	HYDRIC	0.4	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MA01R	VE	BI00329EB	Zinc	HYDRIC	19.4	MG/KG	
MA01R	VE	BI00331EB	Cadmium	HYDRIC	0.4	MG/KG	U
MA01R	VE	BI00331EB	Chromium	HYDRIC	0.6	MG/KG	
MA01R	VE	BI00331EB	Copper	HYDRIC	11.1	MG/KG	
MA01R	VE	BI00331EB	Lead	HYDRIC	0.5	MG/KG	I
MA01R	VE	BI00331EB	Mercury	HYDRIC	0.0	MG/KG	U
MA01R	VE	BI00331EB	Selenium	HYDRIC	0.8	MG/KG	U
MA01R	VE	BI00331EB	Silver	HYDRIC	0.4	MG/KG	U
MA01R	VE	BI00331EB	Zinc	HYDRIC	60.5	MG/KG	
MA02A	SM	BI00173EB	Cadmium	HYDRIC	2.5	MG/KG	U
MA02A	SM	BI00173EB	Cadmium	HYDRIC	2.5	MG/KG	U
MA02A	SM	BI00173EB	Chromium	HYDRIC	3.8	MG/KG	U
MA02A	SM	BI00173EB	Chromium	HYDRIC	3.8	MG/KG	U
MA02A	SM	BI00173EB	Copper	HYDRIC	23.8	MG/KG	
MA02A	SM	BI00173EB	Copper	HYDRIC	18.7	MG/KG	
MA02A	SM	BI00173EB	Lead	HYDRIC	2.3	MG/KG	I
MA02A	SM	BI00173EB	Lead	HYDRIC	1.8	MG/KG	
MA02A	SM	BI00173EB	Mercury	HYDRIC	0.5	MG/KG	U
MA02A	SM	BI00173EB	Mercury	HYDRIC	0.6	MG/KG	U
MA02A	SM	BI00173EB	Selenium	HYDRIC	6.9	MG/KG	
MA02A	SM	BI00173EB	Selenium	HYDRIC	6.0	MG/KG	
MA02A	SM	BI00173EB	Silver	HYDRIC	2.5	MG/KG	U
MA02A	SM	BI00173EB	Silver	HYDRIC	2.5	MG/KG	U
MA02A	SM	BI00173EB	Zinc	HYDRIC	91.1	MG/KG	
MA02A	SM	BI00173EB	Zinc	HYDRIC	98.3	MG/KG	
MA02A	VE	BI00311EB	Cadmium	HYDRIC	0.4	MG/KG	U
MA02A	VE	BI00311EB	Chromium	HYDRIC	37.8	MG/KG	
MA02A	VE	BI00311EB	Copper	HYDRIC	5.1	MG/KG	
MA02A	VE	BI00311EB	Lead	HYDRIC	0.7	MG/KG	
MA02A	VE	BI00311EB	Mercury	HYDRIC	0.0	MG/KG	U
MA02A	VE	BI00311EB	Selenium	HYDRIC	0.9	MG/KG	U
MA02A	VE	BI00311EB	Silver	HYDRIC	0.4	MG/KG	U
MA02A	VE	BI00311EB	Zinc	HYDRIC	7.1	MG/KG	
MA02R	SM	BI00248EB	Cadmium	HYDRIC	1.2	MG/KG	
MA02R	SM	BI00248EB	Chromium	HYDRIC	5.8	MG/KG	U
MA02R	SM	BI00248EB	Copper	HYDRIC	10.8	MG/KG	
MA02R	SM	BI00248EB	Lead	HYDRIC	1.2	MG/KG	
MA02R	SM	BI00248EB	Mercury	HYDRIC	0.5	MG/KG	U
MA02R	SM	BI00248EB	Selenium	HYDRIC	4.7	MG/KG	UI
MA02R	SM	BI00248EB	Silver	HYDRIC	3.5	MG/KG	U
MA02R	SM	BI00248EB	Zinc	HYDRIC	106	MG/KG	
MA03A	SM	BI00214EB	Cadmium	HYDRIC	2.3	MG/KG	U
MA03A	SM	BI00214EB	Chromium	HYDRIC	4.6	MG/KG	
MA03A	SM	BI00214EB	Copper	HYDRIC	18.3	MG/KG	
MA03A	SM	BI00214EB	Lead	HYDRIC	2.0	MG/KG	
MA03A	SM	BI00214EB	Mercury	HYDRIC	0.5	MG/KG	U
MA03A	SM	BI00214EB	Selenium	HYDRIC	4.6	MG/KG	U
MA03A	SM	BI00214EB	Silver	HYDRIC	2.3	MG/KG	U
MA03A	SM	BI00214EB	Zinc	HYDRIC	127	MG/KG	
MA03A	VE	BI00323EB	Cadmium	HYDRIC	0.4	MG/KG	U
MA03A	VE	BI00323EB	Chromium	HYDRIC	0.6	MG/KG	U
MA03A	VE	BI00323EB	Copper	HYDRIC	4.4	MG/KG	
MA03A	VE	BI00323EB	Lead	HYDRIC	0.5	MG/KG	
MA03A	VE	BI00323EB	Mercury	HYDRIC	0.1	MG/KG	U
MA03A	VE	BI00323EB	Selenium	HYDRIC	0.8	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MA03A	VE	BI00323EB	Silver	HYDRIC	0.4	MG/KG	U
MA03A	VE	BI00323EB	Zinc	HYDRIC	7.0	MG/KG	
MA03R	SM	BI00246EB	Cadmium	HYDRIC	1.6	MG/KG	
MA03R	SM	BI00246EB	Chromium	HYDRIC	5.0	MG/KG	U
MA03R	SM	BI00246EB	Copper	HYDRIC	17.0	MG/KG	
MA03R	SM	BI00246EB	Lead	HYDRIC	1.5	MG/KG	
MA03R	SM	BI00246EB	Mercury	HYDRIC	0.5	MG/KG	U
MA03R	SM	BI00246EB	Selenium	HYDRIC	4.0	MG/KG	U
MA03R	SM	BI00246EB	Silver	HYDRIC	3.0	MG/KG	U
MA03R	SM	BI00246EB	Zinc	HYDRIC	96.9	MG/KG	
MA04A	SM	BI00239EB	Cadmium	HYDRIC	1.9	MG/KG	
MA04A	SM	BI00239EB	Chromium	HYDRIC	5.7	MG/KG	U
MA04A	SM	BI00239EB	Copper	HYDRIC	22.0	MG/KG	
MA04A	SM	BI00239EB	Lead	HYDRIC	1.1	MG/KG	U
MA04A	SM	BI00239EB	Mercury	HYDRIC	0.4	MG/KG	U
MA04A	SM	BI00239EB	Selenium	HYDRIC	4.6	MG/KG	UI
MA04A	SM	BI00239EB	Silver	HYDRIC	3.4	MG/KG	U
MA04A	SM	BI00239EB	Zinc	HYDRIC	133	MG/KG	
MA04A	TA	BI00241EB	Cadmium	HYDRIC	6.4	MG/KG	
MA04A	TA	BI00241EB	Chromium	HYDRIC	6.5	MG/KG	U
MA04A	TA	BI00241EB	Copper	HYDRIC	204	MG/KG	
MA04A	TA	BI00241EB	Lead	HYDRIC	3.6	MG/KG	
MA04A	TA	BI00241EB	Mercury	HYDRIC	1.0	MG/KG	U
MA04A	TA	BI00241EB	Selenium	HYDRIC	8.6	MG/KG	UI
MA04A	TA	BI00241EB	Silver	HYDRIC	4.3	MG/KG	U
MA04A	TA	BI00241EB	Zinc	HYDRIC	210	MG/KG	
MA04A	VE	BI00325EB	Cadmium	HYDRIC	0.4	MG/KG	U
MA04A	VE	BI00325EB	Chromium	HYDRIC	1.9	MG/KG	
MA04A	VE	BI00325EB	Copper	HYDRIC	2.8	MG/KG	
MA04A	VE	BI00325EB	Lead	HYDRIC	0.5	MG/KG	
MA04A	VE	BI00325EB	Mercury	HYDRIC	0.1	MG/KG	U
MA04A	VE	BI00325EB	Selenium	HYDRIC	0.8	MG/KG	UI
MA04A	VE	BI00325EB	Silver	HYDRIC	0.4	MG/KG	U
MA04A	VE	BI00325EB	Zinc	HYDRIC	10.1	MG/KG	
MA04R	SM	BI00240EB	Cadmium	HYDRIC	3.1	MG/KG	
MA04R	SM	BI00240EB	Chromium	HYDRIC	6.8	MG/KG	U
MA04R	SM	BI00240EB	Copper	HYDRIC	18.8	MG/KG	
MA04R	SM	BI00240EB	Lead	HYDRIC	1.7	MG/KG	
MA04R	SM	BI00240EB	Mercury	HYDRIC	0.6	MG/KG	U
MA04R	SM	BI00240EB	Selenium	HYDRIC	5.4	MG/KG	U
MA04R	SM	BI00240EB	Silver	HYDRIC	4.1	MG/KG	U
MA04R	SM	BI00240EB	Zinc	HYDRIC	108	MG/KG	
MD01A	SM	BI00190EB	Cadmium	MESIC	3.1	MG/KG	U
MD01A	SM	BI00190EB	Chromium	MESIC	4.6	MG/KG	U
MD01A	SM	BI00190EB	Copper	MESIC	158	MG/KG	
MD01A	SM	BI00190EB	Lead	MESIC	2.6	MG/KG	
MD01A	SM	BI00190EB	Mercury	MESIC	0.6	MG/KG	U
MD01A	SM	BI00190EB	Selenium	MESIC	6.2	MG/KG	UI
MD01A	SM	BI00190EB	Silver	MESIC	3.1	MG/KG	U
MD01A	SM	BI00190EB	Zinc	MESIC	93.8	MG/KG	
MD01A	TA	BI00262EB	Cadmium	MESIC	3.8	MG/KG	
MD01A	TA	BI00262EB	Chromium	MESIC	4.7	MG/KG	U
MD01A	TA	BI00262EB	Copper	MESIC	133	MG/KG	
MD01A	TA	BI00262EB	Lead	MESIC	1.6	MG/KG	U
MD01A	TA	BI00262EB	Mercury	MESIC	0.8	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MD01A	TA	BI00262EB	Selenium	MESIC	6.3	MG/KG	U
MD01A	TA	BI00262EB	Silver	MESIC	3.1	MG/KG	U
MD01A	TA	BI00262EB	Zinc	MESIC	144	MG/KG	
MD01A	VE	BI00308EB	Cadmium	MESIC	0.4	MG/KG	U
MD01A	VE	BI00308EB	Chromium	MESIC	1.5	MG/KG	
MD01A	VE	BI00308EB	Copper	MESIC	4.8	MG/KG	
MD01A	VE	BI00308EB	Lead	MESIC	0.7	MG/KG	I
MD01A	VE	BI00308EB	Mercury	MESIC	0.1	MG/KG	U
MD01A	VE	BI00308EB	Selenium	MESIC	0.8	MG/KG	U
MD01A	VE	BI00308EB	Silver	MESIC	0.4	MG/KG	U
MD01A	VE	BI00308EB	Zinc	MESIC	14.6	MG/KG	
MD01A	VE	BI00310EB	Cadmium	MESIC	0.4	MG/KG	U
MD01A	VE	BI00310EB	Chromium	MESIC	0.6	MG/KG	U
MD01A	VE	BI00310EB	Copper	MESIC	4.4	MG/KG	
MD01A	VE	BI00310EB	Lead	MESIC	0.4	MG/KG	I
MD01A	VE	BI00310EB	Mercury	MESIC	0.0	MG/KG	U
MD01A	VE	BI00310EB	Selenium	MESIC	0.8	MG/KG	U
MD01A	VE	BI00310EB	Silver	MESIC	0.5	MG/KG	
MD01A	VE	BI00310EB	Zinc	MESIC	5.2	MG/KG	
MD01B	SM	BI00269EB	Cadmium	MESIC	3.1	MG/KG	
MD01B	SM	BI00269EB	Chromium	MESIC	6.9	MG/KG	U
MD01B	SM	BI00269EB	Copper	MESIC	26.6	MG/KG	
MD01B	SM	BI00269EB	Lead	MESIC	1.4	MG/KG	U
MD01B	SM	BI00269EB	Mercury	MESIC	0.6	MG/KG	U
MD01B	SM	BI00269EB	Selenium	MESIC	5.5	MG/KG	U
MD01B	SM	BI00269EB	Silver	MESIC	4.1	MG/KG	U
MD01B	SM	BI00269EB	Zinc	MESIC	128	MG/KG	
MD02A	SM	BI00187EB	Cadmium	MESIC	2.1	MG/KG	U
MD02A	SM	BI00187EB	Chromium	MESIC	3.1	MG/KG	U
MD02A	SM	BI00187EB	Copper	MESIC	36.2	MG/KG	
MD02A	SM	BI00187EB	Lead	MESIC	2.3	MG/KG	
MD02A	SM	BI00187EB	Mercury	MESIC	0.5	MG/KG	U
MD02A	SM	BI00187EB	Selenium	MESIC	4.2	MG/KG	U
MD02A	SM	BI00187EB	Silver	MESIC	2.1	MG/KG	U
MD02A	SM	BI00187EB	Zinc	MESIC	113	MG/KG	
MD02A	VE	BI00287EB	Cadmium	MESIC	0.4	MG/KG	U
MD02A	VE	BI00287EB	Chromium	MESIC	0.7	MG/KG	
MD02A	VE	BI00287EB	Copper	MESIC	2.8	MG/KG	
MD02A	VE	BI00287EB	Lead	MESIC	0.3	MG/KG	I
MD02A	VE	BI00287EB	Mercury	MESIC	0.0	MG/KG	U
MD02A	VE	BI00287EB	Selenium	MESIC	0.8	MG/KG	U
MD02A	VE	BI00287EB	Silver	MESIC	0.4	MG/KG	U
MD02A	VE	BI00287EB	Zinc	MESIC	12.7	MG/KG	
MD02A	VE	BI00288EB	Cadmium	MESIC	0.4	MG/KG	U
MD02A	VE	BI00288EB	Chromium	MESIC	0.9	MG/KG	
MD02A	VE	BI00288EB	Copper	MESIC	5.1	MG/KG	
MD02A	VE	BI00288EB	Lead	MESIC	0.4	MG/KG	I
MD02A	VE	BI00288EB	Mercury	MESIC	0.0	MG/KG	U
MD02A	VE	BI00288EB	Selenium	MESIC	0.8	MG/KG	U
MD02A	VE	BI00288EB	Silver	MESIC	0.4	MG/KG	U
MD02A	VE	BI00288EB	Zinc	MESIC	7.8	MG/KG	
MG01A	SM	BI00366EB	Cadmium	MESIC	6.4	MG/KG	
MG01A	SM	BI00366EB	Chromium	MESIC	6.5	MG/KG	U
MG01A	SM	BI00366EB	Copper	MESIC	34.4	MG/KG	
MG01A	SM	BI00366EB	Lead	MESIC	6.9	MG/KG	



## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MG01A	SM	BI00366EB	Mercury	MESIC	0.9	MG/KG	U
MG01A	SM	BI00366EB	Selenium	MESIC	8.7	MG/KG	U
MG01A	SM	BI00366EB	Silver	MESIC	4.3	MG/KG	U
MG01A	SM	BI00366EB	Zinc	MESIC	157	MG/KG	
MG01R	SM	BI00267EB	Cadmium	MESIC	2.4	MG/KG	
MG01R	SM	BI00267EB	Chromium	MESIC	7.3	MG/KG	U
MG01R	SM	BI00267EB	Copper	MESIC	42.8	MG/KG	
MG01R	SM	BI00267EB	Lead	MESIC	1.7	MG/KG	
MG01R	SM	BI00267EB	Mercury	MESIC	0.7	MG/KG	U
MG01R	SM	BI00267EB	Selenium	MESIC	5.9	MG/KG	U
MG01R	SM	BI00267EB	Silver	MESIC	4.4	MG/KG	U
MG01R	SM	BI00267EB	Zinc	MESIC	112	MG/KG	
MG02A	VE	BI00318EB	Cadmium	MESIC	0.4	MG/KG	
MG02A	VE	BI00318EB	Chromium	MESIC	1.2	MG/KG	
MG02A	VE	BI00318EB	Copper	MESIC	9.5	MG/KG	
MG02A	VE	BI00318EB	Lead	MESIC	1.0	MG/KG	
MG02A	VE	BI00318EB	Mercury	MESIC	0.1	MG/KG	U
MG02A	VE	BI00318EB	Selenium	MESIC	0.8	MG/KG	U
MG02A	VE	BI00318EB	Silver	MESIC	0.4	MG/KG	U
MG02A	VE	BI00318EB	Zinc	MESIC	45.6	MG/KG	
MG02A	VE	BI00319EB	Cadmium	MESIC	0.4	MG/KG	U
MG02A	VE	BI00319EB	Chromium	MESIC	1.4	MG/KG	
MG02A	VE	BI00319EB	Copper	MESIC	4.0	MG/KG	
MG02A	VE	BI00319EB	Lead	MESIC	0.8	MG/KG	
MG02A	VE	BI00319EB	Mercury	MESIC	0.1	MG/KG	U
MG02A	VE	BI00319EB	Selenium	MESIC	0.8	MG/KG	U
MG02A	VE	BI00319EB	Silver	MESIC	0.4	MG/KG	U
MG02A	VE	BI00319EB	Zinc	MESIC	14.4	MG/KG	
MG02A	VE	BI00520EB	Cadmium	MESIC	0.6	MG/KG	
MG02A	VE	BI00520EB	Chromium	MESIC	1.0	MG/KG	
MG02A	VE	BI00520EB	Copper	MESIC	11.4	MG/KG	
MG02A	VE	BI00520EB	Lead	MESIC	0.9	MG/KG	
MG02A	VE	BI00520EB	Mercury	MESIC	0.1	MG/KG	U
MG02A	VE	BI00520EB	Selenium	MESIC	0.8	MG/KG	U
MG02A	VE	BI00520EB	Silver	MESIC	0.4	MG/KG	U
MG02A	VE	BI00520EB	Zinc	MESIC	50.9	MG/KG	
MG02A	VE	BI00521EB	Cadmium	MESIC	0.4	MG/KG	U
MG02A	VE	BI00521EB	Chromium	MESIC	1.7	MG/KG	
MG02A	VE	BI00521EB	Copper	MESIC	14.0	MG/KG	
MG02A	VE	BI00521EB	Lead	MESIC	0.7	MG/KG	
MG02A	VE	BI00521EB	Mercury	MESIC	0.0	MG/KG	U
MG02A	VE	BI00521EB	Selenium	MESIC	0.8	MG/KG	U
MG02A	VE	BI00521EB	Silver	MESIC	0.4	MG/KG	U
MG02A	VE	BI00521EB	Zinc	MESIC	14.6	MG/KG	
MG02R	SM	BI00266EB	Cadmium	MESIC	3.0	MG/KG	
MG02R	SM	BI00266EB	Chromium	MESIC	9.1	MG/KG	U
MG02R	SM	BI00266EB	Copper	MESIC	19.6	MG/KG	
MG02R	SM	BI00266EB	Lead	MESIC	1.8	MG/KG	U
MG02R	SM	BI00266EB	Mercury	MESIC	0.8	MG/KG	U
MG02R	SM	BI00266EB	Selenium	MESIC	7.3	MG/KG	U
MG02R	SM	BI00266EB	Silver	MESIC	5.4	MG/KG	U
MG02R	SM	BI00266EB	Zinc	MESIC	144	MG/KG	
MG03A	SM	BI00365EB	Cadmium	MESIC	10.6	MG/KG	
MG03A	SM	BI00365EB	Chromium	MESIC	13.0	MG/KG	U
MG03A	SM	BI00365EB	Copper	MESIC	43.1	MG/KG	

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MG03A	SM	BI00365EB	Lead	MESIC	8.5	MG/KG	
MG03A	SM	BI00365EB	Mercury	MESIC	1.8	MG/KG	U
MG03A	SM	BI00365EB	Selenium	MESIC	17.4	MG/KG	UI
MG03A	SM	BI00365EB	Silver	MESIC	8.7	MG/KG	U
MG03A	SM	BI00365EB	Zinc	MESIC	151	MG/KG	
MG03A	TA	BI00249EB	Cadmium	MESIC	4.6	MG/KG	
MG03A	TA	BI00249EB	Chromium	MESIC	6.5	MG/KG	U
MG03A	TA	BI00249EB	Copper	MESIC	130	MG/KG	
MG03A	TA	BI00249EB	Lead	MESIC	3.5	MG/KG	I
MG03A	TA	BI00249EB	Mercury	MESIC	1.1	MG/KG	U
MG03A	TA	BI00249EB	Selenium	MESIC	8.6	MG/KG	UI
MG03A	TA	BI00249EB	Silver	MESIC	4.3	MG/KG	U
MG03A	TA	BI00249EB	Zinc	MESIC	242	MG/KG	
MG03A	VE	BI00341EB	Cadmium	MESIC	0.4	MG/KG	U
MG03A	VE	BI00341EB	Chromium	MESIC	2.2	MG/KG	
MG03A	VE	BI00341EB	Copper	MESIC	2.0	MG/KG	
MG03A	VE	BI00341EB	Lead	MESIC	0.7	MG/KG	
MG03A	VE	BI00341EB	Mercury	MESIC	0.1	MG/KG	U
MG03A	VE	BI00341EB	Selenium	MESIC	0.8	MG/KG	U
MG03A	VE	BI00341EB	Silver	MESIC	0.4	MG/KG	U
MG03A	VE	BI00341EB	Zinc	MESIC	12.1	MG/KG	
MG03A	VE	BI00517EB	Cadmium	MESIC	1.8	MG/KG	
MG03A	VE	BI00517EB	Chromium	MESIC	2.1	MG/KG	
MG03A	VE	BI00517EB	Copper	MESIC	15.2	MG/KG	
MG03A	VE	BI00517EB	Lead	MESIC	1.1	MG/KG	
MG03A	VE	BI00517EB	Mercury	MESIC	0.0	MG/KG	U
MG03A	VE	BI00517EB	Selenium	MESIC	0.8	MG/KG	U
MG03A	VE	BI00517EB	Silver	MESIC	0.4	MG/KG	U
MG03A	VE	BI00517EB	Zinc	MESIC	48.1	MG/KG	
MG03A	VE	BI00518EB	Cadmium	MESIC	1.9	MG/KG	
MG03A	VE	BI00518EB	Chromium	MESIC	1.2	MG/KG	
MG03A	VE	BI00518EB	Copper	MESIC	14.6	MG/KG	
MG03A	VE	BI00518EB	Lead	MESIC	1.0	MG/KG	
MG03A	VE	BI00518EB	Mercury	MESIC	0.1	MG/KG	U
MG03A	VE	BI00518EB	Selenium	MESIC	0.8	MG/KG	UI
MG03A	VE	BI00518EB	Silver	MESIC	0.4	MG/KG	U
MG03A	VE	BI00518EB	Zinc	MESIC	56.4	MG/KG	
MG03A	VE	BI00519EB	Cadmium	MESIC	0.4	MG/KG	U
MG03A	VE	BI00519EB	Chromium	MESIC	2.1	MG/KG	
MG03A	VE	BI00519EB	Copper	MESIC	2.2	MG/KG	
MG03A	VE	BI00519EB	Lead	MESIC	0.7	MG/KG	
MG03A	VE	BI00519EB	Mercury	MESIC	0.0	MG/KG	U
MG03A	VE	BI00519EB	Selenium	MESIC	0.8	MG/KG	UI
MG03A	VE	BI00519EB	Silver	MESIC	0.4	MG/KG	U
MG03A	VE	BI00519EB	Zinc	MESIC	10.8	MG/KG	
MG03R	TA	BI00301EB	Cadmium	MESIC	3.6	MG/KG	
MG03R	TA	BI00301EB	Chromium	MESIC	4.0	MG/KG	U
MG03R	TA	BI00301EB	Copper	MESIC	73.9	MG/KG	
MG03R	TA	BI00301EB	Lead	MESIC	1.3	MG/KG	U
MG03R	TA	BI00301EB	Mercury	MESIC	0.6	MG/KG	U
MG03R	TA	BI00301EB	Selenium	MESIC	5.4	MG/KG	U
MG03R	TA	BI00301EB	Silver	MESIC	2.7	MG/KG	U
MG03R	TA	BI00301EB	Zinc	MESIC	138	MG/KG	
MG03R	VE	BI00333EB	Cadmium	MESIC	0.4	MG/KG	U
MG03R	VE	BI00333EB	Cadmium	MESIC	0.4	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MG03R	VE	BI00333EB	Chromium	MESIC	1.3	MG/KG	
MG03R	VE	BI00333EB	Chromium	MESIC	1.2	MG/KG	
MG03R	VE	BI00333EB	Copper	MESIC	3.4	MG/KG	
MG03R	VE	BI00333EB	Copper	MESIC	2.9	MG/KG	
MG03R	VE	BI00333EB	Lead	MESIC	1.4	MG/KG	I
MG03R	VE	BI00333EB	Lead	MESIC	1.2	MG/KG	
MG03R	VE	BI00333EB	Mercury	MESIC	0.0	MG/KG	U
MG03R	VE	BI00333EB	Mercury	MESIC	0.0	MG/KG	U
MG03R	VE	BI00333EB	Selenium	MESIC	0.8	MG/KG	U
MG03R	VE	BI00333EB	Selenium	MESIC	0.8	MG/KG	U
MG03R	VE	BI00333EB	Silver	MESIC	0.4	MG/KG	U
MG03R	VE	BI00333EB	Silver	MESIC	0.4	MG/KG	U
MG03R	VE	BI00333EB	Zinc	MESIC	14.3	MG/KG	
MG03R	VE	BI00333EB	Zinc	MESIC	14.2	MG/KG	
MG03R	VE	BI00336EB	Cadmium	MESIC	0.6	MG/KG	
MG03R	VE	BI00336EB	Cadmium	MESIC	84.3	%	
MG03R	VE	BI00336EB	Chromium	MESIC	1.3	MG/KG	
MG03R	VE	BI00336EB	Chromium	MESIC	85.9	%	
MG03R	VE	BI00336EB	Copper	MESIC	10.4	MG/KG	
MG03R	VE	BI00336EB	Copper	MESIC	89.9	%	
MG03R	VE	BI00336EB	Lead	MESIC	1.4	MG/KG	I
MG03R	VE	BI00336EB	Lead	MESIC	73.3	%	
MG03R	VE	BI00336EB	Mercury	MESIC	0.1	MG/KG	U
MG03R	VE	BI00336EB	Mercury	MESIC	63.0	%	U
MG03R	VE	BI00336EB	Selenium	MESIC	1.0	MG/KG	U
MG03R	VE	BI00336EB	Selenium	MESIC	0.7	%	U
MG03R	VE	BI00336EB	Silver	MESIC	0.7	MG/KG	
MG03R	VE	BI00336EB	Silver	MESIC	78.0	%	
MG03R	VE	BI00336EB	Zinc	MESIC	38.3	MG/KG	
MG03R	VE	BI00336EB	Zinc	MESIC	81.4	%	
MG04A	SM	BI00353EB	Cadmium	MESIC	3.0	MG/KG	
MG04A	SM	BI00353EB	Chromium	MESIC	3.3	MG/KG	U
MG04A	SM	BI00353EB	Copper	MESIC	13.0	MG/KG	
MG04A	SM	BI00353EB	Lead	MESIC	1.9	MG/KG	
MG04A	SM	BI00353EB	Mercury	MESIC	0.5	MG/KG	U
MG04A	SM	BI00353EB	Selenium	MESIC	4.4	MG/KG	U
MG04A	SM	BI00353EB	Silver	MESIC	2.2	MG/KG	U
MG04A	SM	BI00353EB	Zinc	MESIC	92.3	MG/KG	
MG04A	VE	BI00255EB	Cadmium	MESIC	0.4	MG/KG	U
MG04A	VE	BI00255EB	Chromium	MESIC	1.6	MG/KG	
MG04A	VE	BI00255EB	Copper	MESIC	1.9	MG/KG	
MG04A	VE	BI00255EB	Lead	MESIC	0.6	MG/KG	
MG04A	VE	BI00255EB	Mercury	MESIC	0.1	MG/KG	U
MG04A	VE	BI00255EB	Selenium	MESIC	0.8	MG/KG	UI
MG04A	VE	BI00255EB	Silver	MESIC	0.4	MG/KG	U
MG04A	VE	BI00255EB	Zinc	MESIC	20.7	MG/KG	
MG04A	VE	BI00256EB	Cadmium	MESIC	0.6	MG/KG	
MG04A	VE	BI00256EB	Chromium	MESIC	1.1	MG/KG	
MG04A	VE	BI00256EB	Copper	MESIC	10.4	MG/KG	
MG04A	VE	BI00256EB	Lead	MESIC	0.9	MG/KG	
MG04A	VE	BI00256EB	Mercury	MESIC	0.1	MG/KG	U
MG04A	VE	BI00256EB	Selenium	MESIC	0.9	MG/KG	U
MG04A	VE	BI00256EB	Silver	MESIC	0.4	MG/KG	U
MG04A	VE	BI00256EB	Zinc	MESIC	46.9	MG/KG	
MG04A	VE	BI00522EB	Cadmium	MESIC	0.4	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MG04A	VE	BI00522EB	Chromium	MESIC	1.3	MG/KG	
MG04A	VE	BI00522EB	Copper	MESIC	5.4	MG/KG	
MG04A	VE	BI00522EB	Lead	MESIC	1.3	MG/KG	
MG04A	VE	BI00522EB	Mercury	MESIC	0.0	MG/KG	U
MG04A	VE	BI00522EB	Selenium	MESIC	0.8	MG/KG	U
MG04A	VE	BI00522EB	Silver	MESIC	0.4	MG/KG	U
MG04A	VE	BI00522EB	Zinc	MESIC	23.7	MG/KG	
MG04A	VE	BI00523EB	Cadmium	MESIC	0.8	MG/KG	
MG04A	VE	BI00523EB	Chromium	MESIC	3.4	MG/KG	
MG04A	VE	BI00523EB	Copper	MESIC	16.3	MG/KG	
MG04A	VE	BI00523EB	Lead	MESIC	1.9	MG/KG	
MG04A	VE	BI00523EB	Mercury	MESIC	0.1	MG/KG	U
MG04A	VE	BI00523EB	Selenium	MESIC	0.8	MG/KG	UI
MG04A	VE	BI00523EB	Silver	MESIC	0.4	MG/KG	U
MG04A	VE	BI00523EB	Zinc	MESIC	59.8	MG/KG	
MG04R	SM	BI00271EB	Cadmium	MESIC	2.7	MG/KG	U
MG04R	SM	BI00271EB	Cadmium	MESIC	3.9	MG/KG	
MG04R	SM	BI00271EB	Chromium	MESIC	4.9	MG/KG	
MG04R	SM	BI00271EB	Chromium	MESIC	4.0	MG/KG	U
MG04R	SM	BI00271EB	Copper	MESIC	16.2	MG/KG	
MG04R	SM	BI00271EB	Copper	MESIC	21.6	MG/KG	
MG04R	SM	BI00271EB	Lead	MESIC	2.7	MG/KG	
MG04R	SM	BI00271EB	Lead	MESIC	2.5	MG/KG	
MG04R	SM	BI00271EB	Mercury	MESIC	0.6	MG/KG	U
MG04R	SM	BI00271EB	Mercury	MESIC	0.6	MG/KG	U
MG04R	SM	BI00271EB	Selenium	MESIC	5.2	MG/KG	U
MG04R	SM	BI00271EB	Selenium	MESIC	5.3	MG/KG	U
MG04R	SM	BI00271EB	Silver	MESIC	2.7	MG/KG	U
MG04R	SM	BI00271EB	Silver	MESIC	2.6	MG/KG	U
MG04R	SM	BI00271EB	Zinc	MESIC	146	MG/KG	
MG04R	SM	BI00271EB	Zinc	MESIC	124	MG/KG	
MG04R	VE	BI00280EB	Cadmium	MESIC	1.2	MG/KG	
MG04R	VE	BI00280EB	Chromium	MESIC	1.1	MG/KG	
MG04R	VE	BI00280EB	Copper	MESIC	10.7	MG/KG	
MG04R	VE	BI00280EB	Lead	MESIC	1.1	MG/KG	I
MG04R	VE	BI00280EB	Mercury	MESIC	0.0	MG/KG	U
MG04R	VE	BI00280EB	Selenium	MESIC	0.8	MG/KG	UI
MG04R	VE	BI00280EB	Silver	MESIC	0.4	MG/KG	U
MG04R	VE	BI00280EB	Zinc	MESIC	44.8	MG/KG	
MG04R	VE	BI00281EB	Cadmium	MESIC	0.4	MG/KG	U
MG04R	VE	BI00281EB	Chromium	MESIC	1.2	MG/KG	
MG04R	VE	BI00281EB	Copper	MESIC	4.7	MG/KG	
MG04R	VE	BI00281EB	Lead	MESIC	0.7	MG/KG	I
MG04R	VE	BI00281EB	Mercury	MESIC	0.0	MG/KG	U
MG04R	VE	BI00281EB	Selenium	MESIC	0.8	MG/KG	U
MG04R	VE	BI00281EB	Silver	MESIC	0.4	MG/KG	U
MG04R	VE	BI00281EB	Zinc	MESIC	19.8	MG/KG	
MR01A	VE	BI00363EB	Cadmium	MESIC	0.4	MG/KG	U
MR01A	VE	BI00363EB	Chromium	MESIC	1.5	MG/KG	
MR01A	VE	BI00363EB	Copper	MESIC	2.2	MG/KG	
MR01A	VE	BI00363EB	Lead	MESIC	0.6	MG/KG	
MR01A	VE	BI00363EB	Mercury	MESIC	0.0	MG/KG	U
MR01A	VE	BI00363EB	Selenium	MESIC	0.8	MG/KG	UI
MR01A	VE	BI00363EB	Silver	MESIC	0.4	MG/KG	U
MR01A	VE	BI00363EB	Zinc	MESIC	14.0	MG/KG	

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MR02A	SM	BI00364EB	Cadmium	MESIC	4.1	MG/KG	
MR02A	SM	BI00364EB	Chromium	MESIC	5.1	MG/KG	U
MR02A	SM	BI00364EB	Copper	MESIC	18.2	MG/KG	
MR02A	SM	BI00364EB	Lead	MESIC	3.1	MG/KG	
MR02A	SM	BI00364EB	Mercury	MESIC	0.8	MG/KG	U
MR02A	SM	BI00364EB	Selenium	MESIC	6.8	MG/KG	U
MR02A	SM	BI00364EB	Silver	MESIC	3.4	MG/KG	U
MR02A	SM	BI00364EB	Zinc	MESIC	139	MG/KG	
MR02A	VE	BI00320EB	Cadmium	MESIC	0.5	MG/KG	
MR02A	VE	BI00320EB	Chromium	MESIC	2.7	MG/KG	
MR02A	VE	BI00320EB	Copper	MESIC	12.6	MG/KG	
MR02A	VE	BI00320EB	Lead	MESIC	1.0	MG/KG	
MR02A	VE	BI00320EB	Mercury	MESIC	0.0	MG/KG	U
MR02A	VE	BI00320EB	Selenium	MESIC	0.8	MG/KG	UI
MR02A	VE	BI00320EB	Silver	MESIC	0.4	MG/KG	U
MR02A	VE	BI00320EB	Zinc	MESIC	23.1	MG/KG	
MR03A	TA	BI00302EB	Cadmium	MESIC	3.6	MG/KG	U
MR03A	TA	BI00302EB	Chromium	MESIC	5.4	MG/KG	U
MR03A	TA	BI00302EB	Copper	MESIC	89.2	MG/KG	
MR03A	TA	BI00302EB	Lead	MESIC	1.9	MG/KG	
MR03A	TA	BI00302EB	Mercury	MESIC	0.8	MG/KG	U
MR03A	TA	BI00302EB	Selenium	MESIC	7.3	MG/KG	U
MR03A	TA	BI00302EB	Silver	MESIC	3.6	MG/KG	U
MR03A	TA	BI00302EB	Zinc	MESIC	162	MG/KG	
MR03A	VE	BI00282EB	Cadmium	MESIC	0.9	MG/KG	
MR03A	VE	BI00282EB	Cadmium	MESIC	0.8	MG/KG	
MR03A	VE	BI00282EB	Chromium	MESIC	1.2	MG/KG	
MR03A	VE	BI00282EB	Chromium	MESIC	1.6	MG/KG	
MR03A	VE	BI00282EB	Copper	MESIC	10.4	MG/KG	
MR03A	VE	BI00282EB	Copper	MESIC	9.7	MG/KG	
MR03A	VE	BI00282EB	Lead	MESIC	0.7	MG/KG	
MR03A	VE	BI00282EB	Lead	MESIC	0.6	MG/KG	
MR03A	VE	BI00282EB	Mercury	MESIC	0.1	MG/KG	U
MR03A	VE	BI00282EB	Selenium	MESIC	1.5	MG/KG	
MR03A	VE	BI00282EB	Selenium	MESIC	0.9	MG/KG	
MR03A	VE	BI00282EB	Silver	MESIC	0.4	MG/KG	U
MR03A	VE	BI00282EB	Silver	MESIC	0.4	MG/KG	U
MR03A	VE	BI00282EB	Zinc	MESIC	43.3	MG/KG	
MR03A	VE	BI00282EB	Zinc	MESIC	40.9	MG/KG	
MR03A	VE	BI00283EB	Cadmium	MESIC	0.3	MG/KG	U
MR03A	VE	BI00283EB	Cadmium	MESIC	86.2	%	U
MR03A	VE	BI00283EB	Chromium	MESIC	1.0	MG/KG	
MR03A	VE	BI00283EB	Chromium	MESIC	91.8	%	
MR03A	VE	BI00283EB	Copper	MESIC	2.0	MG/KG	
MR03A	VE	BI00283EB	Copper	MESIC	92.7	%	
MR03A	VE	BI00283EB	Lead	MESIC	0.2	MG/KG	UI
MR03A	VE	BI00283EB	Lead	MESIC	93.8	%	U
MR03A	VE	BI00283EB	Mercury	MESIC	0.1	MG/KG	U
MR03A	VE	BI00283EB	Selenium	MESIC	5.3	MG/KG	
MR03A	VE	BI00283EB	Selenium	MESIC	-167	%	
MR03A	VE	BI00283EB	Silver	MESIC	0.3	MG/KG	U
MR03A	VE	BI00283EB	Silver	MESIC	80.5	%	U
MR03A	VE	BI00283EB	Zinc	MESIC	6.1	MG/KG	
MR03A	VE	BI00283EB	Zinc	MESIC	87.1	%	
MR04A	SM	BI00354EB	Cadmium	MESIC	2.6	MG/KG	

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MR04A	SM	BI00354EB	Chromium	MESIC	3.6	MG/KG	U
MR04A	SM	BI00354EB	Copper	MESIC	18.6	MG/KG	
MR04A	SM	BI00354EB	Lead	MESIC	1.5	MG/KG	
MR04A	SM	BI00354EB	Mercury	MESIC	0.6	MG/KG	U
MR04A	SM	BI00354EB	Selenium	MESIC	4.8	MG/KG	U
MR04A	SM	BI00354EB	Silver	MESIC	2.4	MG/KG	U
MR04A	SM	BI00354EB	Zinc	MESIC	91.2	MG/KG	
MR04A	VE	BI00259EB	Cadmium	MESIC	0.4	MG/KG	U
MR04A	VE	BI00259EB	Chromium	MESIC	1.9	MG/KG	
MR04A	VE	BI00259EB	Copper	MESIC	10	MG/KG	
MR04A	VE	BI00259EB	Lead	MESIC	0.9	MG/KG	I
MR04A	VE	BI00259EB	Mercury	MESIC	0.0	MG/KG	U
MR04A	VE	BI00259EB	Selenium	MESIC	0.8	MG/KG	U
MR04A	VE	BI00259EB	Silver	MESIC	0.4	MG/KG	U
MR04A	VE	BI00259EB	Zinc	MESIC	29.1	MG/KG	
MR04A	VE	BI00261EB	Cadmium	MESIC	0.4	MG/KG	U
MR04A	VE	BI00261EB	Chromium	MESIC	1.5	MG/KG	
MR04A	VE	BI00261EB	Copper	MESIC	1.8	MG/KG	
MR04A	VE	BI00261EB	Lead	MESIC	0.2	MG/KG	UI
MR04A	VE	BI00261EB	Mercury	MESIC	0.1	MG/KG	U
MR04A	VE	BI00261EB	Selenium	MESIC	0.8	MG/KG	U
MR04A	VE	BI00261EB	Silver	MESIC	0.4	MG/KG	U
MR04A	VE	BI00261EB	Zinc	MESIC	24.0	MG/KG	
MR04A	VE	BI00515EB	Cadmium	MESIC	0.4	MG/KG	U
MR04A	VE	BI00515EB	Chromium	MESIC	2.0	MG/KG	
MR04A	VE	BI00515EB	Copper	MESIC	1.9	MG/KG	
MR04A	VE	BI00515EB	Lead	MESIC	0.2	MG/KG	I
MR04A	VE	BI00515EB	Mercury	MESIC	0.1	MG/KG	U
MR04A	VE	BI00515EB	Mercury	MESIC	0.1	MG/KG	U
MR04A	VE	BI00515EB	Selenium	MESIC	3.1	MG/KG	
MR04A	VE	BI00515EB	Silver	MESIC	0.4	MG/KG	U
MR04A	VE	BI00515EB	Zinc	MESIC	22.5	MG/KG	
MR04A	VE	BI00516EB	Cadmium	MESIC	0.5	MG/KG	
MR04A	VE	BI00516EB	Chromium	MESIC	1.6	MG/KG	
MR04A	VE	BI00516EB	Copper	MESIC	8.1	MG/KG	
MR04A	VE	BI00516EB	Lead	MESIC	0.7	MG/KG	I
MR04A	VE	BI00516EB	Mercury	MESIC	0.1	MG/KG	U
MR04A	VE	BI00516EB	Mercury	MESIC	77.0	%	U
MR04A	VE	BI00516EB	Selenium	MESIC	2.3	MG/KG	I
MR04A	VE	BI00516EB	Silver	MESIC	0.4	MG/KG	U
MR04A	VE	BI00516EB	Zinc	MESIC	30.0	MG/KG	
MW01A	SM	BI00189EB	Cadmium	HYDRIC	2.6	MG/KG	U
MW01A	SM	BI00189EB	Chromium	HYDRIC	3.9	MG/KG	U
MW01A	SM	BI00189EB	Copper	HYDRIC	15.7	MG/KG	
MW01A	SM	BI00189EB	Lead	HYDRIC	2.1	MG/KG	
MW01A	SM	BI00189EB	Mercury	HYDRIC	0.6	MG/KG	U
MW01A	SM	BI00189EB	Selenium	HYDRIC	5.2	MG/KG	U
MW01A	SM	BI00189EB	Silver	HYDRIC	2.6	MG/KG	U
MW01A	SM	BI00189EB	Zinc	HYDRIC	210	MG/KG	
MW01A	TA	BI00300EB	Cadmium	HYDRIC	0.7	MG/KG	U
MW01A	TA	BI00300EB	Chromium	HYDRIC	3.5	MG/KG	U
MW01A	TA	BI00300EB	Copper	HYDRIC	32.1	MG/KG	
MW01A	TA	BI00300EB	Lead	HYDRIC	0.9	MG/KG	
MW01A	TA	BI00300EB	Mercury	HYDRIC	0.3	MG/KG	U
MW01A	TA	BI00300EB	Selenium	HYDRIC	2.8	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MW01A	TA	BI00300EB	Silver	HYDRIC	2.1	MG/KG	U
MW01A	TA	BI00300EB	Zinc	HYDRIC	47.0	MG/KG	
MW01A	VE	BI00293EB	Cadmium	HYDRIC	0.3	MG/KG	
MW01A	VE	BI00293EB	Chromium	HYDRIC	34.1	MG/KG	
MW01A	VE	BI00293EB	Copper	HYDRIC	2.1	MG/KG	
MW01A	VE	BI00293EB	Lead	HYDRIC	0.8	MG/KG	
MW01A	VE	BI00293EB	Mercury	HYDRIC	0.0	MG/KG	U
MW01A	VE	BI00293EB	Selenium	HYDRIC	0.9	MG/KG	UI
MW01A	VE	BI00293EB	Silver	HYDRIC	0.6	MG/KG	U
MW01A	VE	BI00293EB	Zinc	HYDRIC	28.1	MG/KG	
MW01R	SM	BI00242EB	Cadmium	HYDRIC	1.4	MG/KG	U
MW01R	SM	BI00242EB	Chromium	HYDRIC	6.8	MG/KG	U
MW01R	SM	BI00242EB	Copper	HYDRIC	16.8	MG/KG	
MW01R	SM	BI00242EB	Lead	HYDRIC	2.6	MG/KG	
MW01R	SM	BI00242EB	Mercury	HYDRIC	0.5	MG/KG	U
MW01R	SM	BI00242EB	Selenium	HYDRIC	5.4	MG/KG	U
MW01R	SM	BI00242EB	Silver	HYDRIC	4.1	MG/KG	U
MW01R	SM	BI00242EB	Zinc	HYDRIC	106	MG/KG	
MW02A	SM	BI00232EB	Cadmium	HYDRIC	2.3	MG/KG	
MW02A	SM	BI00232EB	Chromium	HYDRIC	6.9	MG/KG	U
MW02A	SM	BI00232EB	Copper	HYDRIC	21.3	MG/KG	
MW02A	SM	BI00232EB	Lead	HYDRIC	1.8	MG/KG	
MW02A	SM	BI00232EB	Mercury	HYDRIC	0.6	MG/KG	U
MW02A	SM	BI00232EB	Selenium	HYDRIC	5.5	MG/KG	U
MW02A	SM	BI00232EB	Silver	HYDRIC	4.1	MG/KG	U
MW02A	SM	BI00232EB	Zinc	HYDRIC	104	MG/KG	
MW02R	SM	BI00243EB	Cadmium	HYDRIC	1.3	MG/KG	U
MW02R	SM	BI00243EB	Chromium	HYDRIC	6.3	MG/KG	U
MW02R	SM	BI00243EB	Copper	HYDRIC	16.6	MG/KG	
MW02R	SM	BI00243EB	Lead	HYDRIC	1.6	MG/KG	
MW02R	SM	BI00243EB	Mercury	HYDRIC	0.5	MG/KG	U
MW02R	SM	BI00243EB	Selenium	HYDRIC	5.1	MG/KG	U
MW02R	SM	BI00243EB	Silver	HYDRIC	3.8	MG/KG	U
MW02R	SM	BI00243EB	Zinc	HYDRIC	101	MG/KG	
MW03A	HE	BI00174EB	Cadmium	HYDRIC	2.5	MG/KG	
MW03A	HE	BI00174EB	Cadmium	HYDRIC	2.3	MG/KG	
MW03A	HE	BI00174EB	Chromium	HYDRIC	4.2	MG/KG	U
MW03A	HE	BI00174EB	Chromium	HYDRIC	4.2	MG/KG	U
MW03A	HE	BI00174EB	Copper	HYDRIC	15.7	MG/KG	
MW03A	HE	BI00174EB	Copper	HYDRIC	17.4	MG/KG	
MW03A	HE	BI00174EB	Lead	HYDRIC	2.4	MG/KG	
MW03A	HE	BI00174EB	Lead	HYDRIC	2.5	MG/KG	
MW03A	HE	BI00174EB	Mercury	HYDRIC	0.4	MG/KG	U
MW03A	HE	BI00174EB	Mercury	HYDRIC	0.4	MG/KG	U
MW03A	HE	BI00174EB	Selenium	HYDRIC	11.4	MG/KG	
MW03A	HE	BI00174EB	Selenium	HYDRIC	9.3	MG/KG	
MW03A	HE	BI00174EB	Silver	HYDRIC	2.5	MG/KG	U
MW03A	HE	BI00174EB	Silver	HYDRIC	2.5	MG/KG	U
MW03A	HE	BI00174EB	Zinc	HYDRIC	85.8	MG/KG	
MW03A	HE	BI00174EB	Zinc	HYDRIC	129	MG/KG	
MW03A	SM	BI00188EB	Cadmium	HYDRIC	2.3	MG/KG	U
MW03A	SM	BI00188EB	Chromium	HYDRIC	3.4	MG/KG	
MW03A	SM	BI00188EB	Copper	HYDRIC	17.3	MG/KG	
MW03A	SM	BI00188EB	Lead	HYDRIC	1.2	MG/KG	
MW03A	SM	BI00188EB	Mercury	HYDRIC	0.4	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MW03A	SM	BI00188EB	Selenium	HYDRIC	4.5	MG/KG	UI
MW03A	SM	BI00188EB	Silver	HYDRIC	2.3	MG/KG	U
MW03A	SM	BI00188EB	Zinc	HYDRIC	57.8	MG/KG	
MW03A	VE	BI00312EB	Cadmium	HYDRIC	0.2	MG/KG	
MW03A	VE	BI00312EB	Chromium	HYDRIC	1.5	MG/KG	
MW03A	VE	BI00312EB	Copper	HYDRIC	1.7	MG/KG	
MW03A	VE	BI00312EB	Lead	HYDRIC	0.6	MG/KG	I
MW03A	VE	BI00312EB	Mercury	HYDRIC	0.0	MG/KG	U
MW03A	VE	BI00312EB	Selenium	HYDRIC	0.8	MG/KG	UI
MW03A	VE	BI00312EB	Silver	HYDRIC	0.6	MG/KG	U
MW03A	VE	BI00312EB	Zinc	HYDRIC	13.0	MG/KG	
MW03A	VE	BI00313EB	Cadmium	HYDRIC	1.9	MG/KG	
MW03A	VE	BI00313EB	Cadmium	HYDRIC	1.6	MG/KG	
MW03A	VE	BI00313EB	Chromium	HYDRIC	2.5	MG/KG	
MW03A	VE	BI00313EB	Chromium	HYDRIC	3.0	MG/KG	
MW03A	VE	BI00313EB	Copper	HYDRIC	11.5	MG/KG	
MW03A	VE	BI00313EB	Copper	HYDRIC	11.2	MG/KG	
MW03A	VE	BI00313EB	Lead	HYDRIC	0.8	MG/KG	
MW03A	VE	BI00313EB	Lead	HYDRIC	0.7	MG/KG	
MW03A	VE	BI00313EB	Mercury	HYDRIC	0.1	MG/KG	U
MW03A	VE	BI00313EB	Mercury	HYDRIC	0.1	MG/KG	U
MW03A	VE	BI00313EB	Selenium	HYDRIC	0.8	MG/KG	U
MW03A	VE	BI00313EB	Selenium	HYDRIC	4.3	MG/KG	U
MW03A	VE	BI00313EB	Silver	HYDRIC	0.6	MG/KG	U
MW03A	VE	BI00313EB	Silver	HYDRIC	0.6	MG/KG	U
MW03A	VE	BI00313EB	Zinc	HYDRIC	53.0	MG/KG	
MW03A	VE	BI00313EB	Zinc	HYDRIC	53.3	MG/KG	
MW03R	SM	BI00244EB	Cadmium	HYDRIC	0.9	MG/KG	
MW03R	SM	BI00244EB	Chromium	HYDRIC	4.7	MG/KG	U
MW03R	SM	BI00244EB	Copper	HYDRIC	11.5	MG/KG	
MW03R	SM	BI00244EB	Lead	HYDRIC	0.9	MG/KG	U
MW03R	SM	BI00244EB	Mercury	HYDRIC	0.3	MG/KG	U
MW03R	SM	BI00244EB	Selenium	HYDRIC	3.7	MG/KG	UI
MW03R	SM	BI00244EB	Silver	HYDRIC	2.8	MG/KG	U
MW03R	SM	BI00244EB	Zinc	HYDRIC	74.5	MG/KG	
MW03R	TA	BI00327EB	Cadmium	HYDRIC	3.6	MG/KG	
MW03R	TA	BI00327EB	Chromium	HYDRIC	5.0	MG/KG	U
MW03R	TA	BI00327EB	Copper	HYDRIC	66.3	MG/KG	
MW03R	TA	BI00327EB	Lead	HYDRIC	3.2	MG/KG	
MW03R	TA	BI00327EB	Mercury	HYDRIC	0.7	MG/KG	U
MW03R	TA	BI00327EB	Selenium	HYDRIC	6.6	MG/KG	U
MW03R	TA	BI00327EB	Silver	HYDRIC	3.3	MG/KG	U
MW03R	TA	BI00327EB	Zinc	HYDRIC	129	MG/KG	
MW03R	VE	BI00304EB	Cadmium	HYDRIC	1.1	MG/KG	
MW03R	VE	BI00304EB	Chromium	HYDRIC	5.4	MG/KG	
MW03R	VE	BI00304EB	Copper	HYDRIC	10.2	MG/KG	
MW03R	VE	BI00304EB	Lead	HYDRIC	0.9	MG/KG	
MW03R	VE	BI00304EB	Mercury	HYDRIC	0.1	MG/KG	U
MW03R	VE	BI00304EB	Selenium	HYDRIC	0.9	MG/KG	U
MW03R	VE	BI00304EB	Silver	HYDRIC	0.4	MG/KG	U
MW03R	VE	BI00304EB	Zinc	HYDRIC	55.9	MG/KG	
MW03R	VE	BI00305EB	Cadmium	HYDRIC	0.2	MG/KG	U
MW03R	VE	BI00305EB	Chromium	HYDRIC	1.5	MG/KG	
MW03R	VE	BI00305EB	Copper	HYDRIC	1.7	MG/KG	
MW03R	VE	BI00305EB	Lead	HYDRIC	0.6	MG/KG	



## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MW03R	VE	BI00305EB	Mercury	HYDRIC	0.0	MG/KG	U
MW03R	VE	BI00305EB	Selenium	HYDRIC	0.8	MG/KG	UI
MW03R	VE	BI00305EB	Silver	HYDRIC	0.6	MG/KG	U
MW03R	VE	BI00305EB	Zinc	HYDRIC	12.6	MG/KG	
MW04A	SM	BI00186EB	Cadmium	HYDRIC	2.2	MG/KG	U
MW04A	SM	BI00186EB	Cadmium	HYDRIC	95.7	%	U
MW04A	SM	BI00186EB	Chromium	HYDRIC	3.3	MG/KG	U
MW04A	SM	BI00186EB	Chromium	HYDRIC	94.5	%	U
MW04A	SM	BI00186EB	Copper	HYDRIC	93.6	%	
MW04A	SM	BI00186EB	Copper	HYDRIC	28.2	MG/KG	
MW04A	SM	BI00186EB	Lead	HYDRIC	2.9	MG/KG	
MW04A	SM	BI00186EB	Lead	HYDRIC	92.1	%	
MW04A	SM	BI00186EB	Mercury	HYDRIC	85.8	%	U
MW04A	SM	BI00186EB	Mercury	HYDRIC	0.5	MG/KG	U
MW04A	SM	BI00186EB	Selenium	HYDRIC	4.4	MG/KG	UI
MW04A	SM	BI00186EB	Selenium	HYDRIC	135	%	U
MW04A	SM	BI00186EB	Silver	HYDRIC	88.0	%	U
MW04A	SM	BI00186EB	Silver	HYDRIC	2.2	MG/KG	U
MW04A	SM	BI00186EB	Zinc	HYDRIC	153	MG/KG	
MW04A	SM	BI00186EB	Zinc	HYDRIC	89.5	%	
MW04A	VE	BI00344EB	Cadmium	HYDRIC	0.2	MG/KG	U
MW04A	VE	BI00344EB	Chromium	HYDRIC	0.9	MG/KG	U
MW04A	VE	BI00344EB	Copper	HYDRIC	1.6	MG/KG	
MW04A	VE	BI00344EB	Lead	HYDRIC	0.4	MG/KG	
MW04A	VE	BI00344EB	Mercury	HYDRIC	0.1	MG/KG	U
MW04A	VE	BI00344EB	Selenium	HYDRIC	0.7	MG/KG	UI
MW04A	VE	BI00344EB	Silver	HYDRIC	0.5	MG/KG	U
MW04A	VE	BI00344EB	Zinc	HYDRIC	14.2	MG/KG	
MW04A	VE	BI00345EB	Cadmium	HYDRIC	1.1	MG/KG	
MW04A	VE	BI00345EB	Chromium	HYDRIC	23.3	MG/KG	
MW04A	VE	BI00345EB	Copper	HYDRIC	7.5	MG/KG	
MW04A	VE	BI00345EB	Lead	HYDRIC	0.9	MG/KG	I
MW04A	VE	BI00345EB	Mercury	HYDRIC	0.1	MG/KG	U
MW04A	VE	BI00345EB	Selenium	HYDRIC	0.8	MG/KG	UI
MW04A	VE	BI00345EB	Silver	HYDRIC	0.6	MG/KG	U
MW04A	VE	BI00345EB	Zinc	HYDRIC	53.9	MG/KG	
MX01R	HE	BI00299EB	Cadmium	XERIC	0.6	MG/KG	U
MX01R	HE	BI00299EB	Chromium	XERIC	3.1	MG/KG	U
MX01R	HE	BI00299EB	Copper	XERIC	6.4	MG/KG	
MX01R	HE	BI00299EB	Lead	XERIC	1.1	MG/KG	
MX01R	HE	BI00299EB	Mercury	XERIC	0.2	MG/KG	U
MX01R	HE	BI00299EB	Selenium	XERIC	2.5	MG/KG	UI
MX01R	HE	BI00299EB	Silver	XERIC	1.9	MG/KG	U
MX01R	HE	BI00299EB	Zinc	XERIC	32.5	MG/KG	
MX01R	SM	BI00268EB	Cadmium	XERIC	1.9	MG/KG	
MX01R	SM	BI00268EB	Chromium	XERIC	7.0	MG/KG	U
MX01R	SM	BI00268EB	Copper	XERIC	19.5	MG/KG	
MX01R	SM	BI00268EB	Lead	XERIC	1.5	MG/KG	
MX01R	SM	BI00268EB	Mercury	XERIC	0.6	MG/KG	U
MX01R	SM	BI00268EB	Selenium	XERIC	5.6	MG/KG	U
MX01R	SM	BI00268EB	Silver	XERIC	4.2	MG/KG	U
MX01R	SM	BI00268EB	Zinc	XERIC	113	MG/KG	
MX02R	SM	BI00263EB	Cadmium	XERIC	1.0	MG/KG	U
MX02R	SM	BI00263EB	Chromium	XERIC	5.1	MG/KG	U
MX02R	SM	BI00263EB	Copper	XERIC	16.6	MG/KG	

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MX02R	SM	BI00263EB	Lead	XERIC	1.2	MG/KG	
MX02R	SM	BI00263EB	Mercury	XERIC	0.4	MG/KG	U
MX02R	SM	BI00263EB	Selenium	XERIC	4.1	MG/KG	U
MX02R	SM	BI00263EB	Silver	XERIC	3.1	MG/KG	U
MX02R	SM	BI00263EB	Zinc	XERIC	109	MG/KG	
MX03R	SM	BI00265EB	Cadmium	XERIC	3.3	MG/KG	
MX03R	SM	BI00265EB	Chromium	XERIC	7.3	MG/KG	U
MX03R	SM	BI00265EB	Copper	XERIC	22.5	MG/KG	
MX03R	SM	BI00265EB	Lead	XERIC	1.7	MG/KG	
MX03R	SM	BI00265EB	Mercury	XERIC	0.7	MG/KG	U
MX03R	SM	BI00265EB	Selenium	XERIC	5.8	MG/KG	UI
MX03R	SM	BI00265EB	Silver	XERIC	4.4	MG/KG	U
MX03R	SM	BI00265EB	Zinc	XERIC	117	MG/KG	
RCSP01	FI	BI00504EB	Cadmium	AQAUTIC	1.9	MG/KG	U
RCSP01	FI	BI00504EB	Chromium	AQAUTIC	4.3	MG/KG	
RCSP01	FI	BI00504EB	Copper	AQAUTIC	14.5	MG/KG	
RCSP01	FI	BI00504EB	Lead	AQAUTIC	0.9	MG/KG	U
RCSP01	FI	BI00504EB	Mercury	AQAUTIC	0.4	MG/KG	U
RCSP01	FI	BI00504EB	Selenium	AQAUTIC	18.8	MG/KG	UI
RCSP01	FI	BI00504EB	Silver	AQAUTIC	1.9	MG/KG	U
RCSP01	FI	BI00504EB	Zinc	AQAUTIC	315	MG/KG	
RCSP01	FI	BI00505EB	Cadmium	AQAUTIC	3.9	MG/KG	U
RCSP01	FI	BI00505EB	Chromium	AQAUTIC	5.8	MG/KG	U
RCSP01	FI	BI00505EB	Copper	AQAUTIC	36.8	MG/KG	
RCSP01	FI	BI00505EB	Lead	AQAUTIC	3.0	MG/KG	
RCSP01	FI	BI00505EB	Mercury	AQAUTIC	0.9	MG/KG	U
RCSP01	FI	BI00505EB	Selenium	AQAUTIC	38.8	MG/KG	UI
RCSP01	FI	BI00505EB	Silver	AQAUTIC	3.9	MG/KG	U
RCSP01	FI	BI00505EB	Zinc	AQAUTIC	500	MG/KG	
RCSP01	FI	BI00506EB	Cadmium	AQAUTIC	4.1	MG/KG	U
RCSP01	FI	BI00506EB	Chromium	AQAUTIC	7.4	MG/KG	
RCSP01	FI	BI00506EB	Copper	AQAUTIC	40.9	MG/KG	
RCSP01	FI	BI00506EB	Lead	AQAUTIC	4.3	MG/KG	
RCSP01	FI	BI00506EB	Mercury	AQAUTIC	0.9	MG/KG	U
RCSP01	FI	BI00506EB	Selenium	AQAUTIC	40.8	MG/KG	UI
RCSP01	FI	BI00506EB	Silver	AQAUTIC	4.1	MG/KG	U
RCSP01	FI	BI00506EB	Zinc	AQAUTIC	648	MG/KG	
SW003	HE	BI00185EB	Cadmium	AQAUTIC	0.8	MG/KG	U
SW003	HE	BI00185EB	Cadmium	AQAUTIC	94.5	%	U
SW003	HE	BI00185EB	Chromium	AQAUTIC	89.7	%	U
SW003	HE	BI00185EB	Chromium	AQAUTIC	4.4	MG/KG	U
SW003	HE	BI00185EB	Copper	AQAUTIC	3.5	MG/KG	U
SW003	HE	BI00185EB	Copper	AQAUTIC	95.5	%	U
SW003	HE	BI00185EB	Lead	AQAUTIC	261	%	
SW003	HE	BI00185EB	Lead	AQAUTIC	1.3	MG/KG	
SW003	HE	BI00185EB	Mercury	AQAUTIC	0.4	MG/KG	U
SW003	HE	BI00185EB	Mercury	AQAUTIC	74.0	%	U
SW003	HE	BI00185EB	Selenium	AQAUTIC	135	%	U
SW003	HE	BI00185EB	Selenium	AQAUTIC	3.5	MG/KG	UI
SW003	HE	BI00185EB	Silver	AQAUTIC	2.6	MG/KG	U
SW003	HE	BI00185EB	Silver	AQAUTIC	85.7	%	U
SW003	HE	BI00185EB	Zinc	AQAUTIC	94.0	%	
SW003	HE	BI00185EB	Zinc	AQAUTIC	19.5	MG/KG	
SW005	FI	BI00275EB	Cadmium	AQAUTIC	2.4	MG/KG	
SW005	FI	BI00275EB	Cadmium	AQAUTIC	89.8	%	

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SW005	FI	BI00275EB	Chromium	AQAUTIC	2.9	MG/KG	U
SW005	FI	BI00275EB	Chromium	AQAUTIC	94.6	%	U
SW005	FI	BI00275EB	Copper	AQAUTIC	10.3	MG/KG	
SW005	FI	BI00275EB	Copper	AQAUTIC	95.4	%	
SW005	FI	BI00275EB	Lead	AQAUTIC	1.9	MG/KG	
SW005	FI	BI00275EB	Lead	AQAUTIC	90.6	%	
SW005	FI	BI00275EB	Mercury	AQAUTIC	0.4	MG/KG	U
SW005	FI	BI00275EB	Mercury	AQAUTIC	90.8	%	U
SW005	FI	BI00275EB	Selenium	AQAUTIC	3.9	MG/KG	U
SW005	FI	BI00275EB	Selenium	AQAUTIC	54.7	%	U
SW005	FI	BI00275EB	Silver	AQAUTIC	1.9	MG/KG	U
SW005	FI	BI00275EB	Silver	AQAUTIC	91.4	%	U
SW005	FI	BI00275EB	Zinc	AQAUTIC	78.1	MG/KG	
SW005	FI	BI00275EB	Zinc	AQAUTIC	98.9	%	
SW005	FI	BI00276EB	Cadmium	AQAUTIC	1.6	MG/KG	U
SW005	FI	BI00276EB	Chromium	AQAUTIC	2.5	MG/KG	U
SW005	FI	BI00276EB	Copper	AQAUTIC	3.6	MG/KG	
SW005	FI	BI00276EB	Lead	AQAUTIC	0.8	MG/KG	
SW005	FI	BI00276EB	Mercury	AQAUTIC	0.5	MG/KG	
SW005	FI	BI00276EB	Selenium	AQAUTIC	3.3	MG/KG	UI
SW005	FI	BI00276EB	Silver	AQAUTIC	1.6	MG/KG	U
SW005	FI	BI00276EB	Zinc	AQAUTIC	79.9	MG/KG	
SW005	FI	BI00277EB	Cadmium	AQAUTIC	2.5	MG/KG	
SW005	FI	BI00277EB	Chromium	AQAUTIC	3.8	MG/KG	
SW005	FI	BI00277EB	Copper	AQAUTIC	6.8	MG/KG	
SW005	FI	BI00277EB	Lead	AQAUTIC	1.7	MG/KG	
SW005	FI	BI00277EB	Mercury	AQAUTIC	0.5	MG/KG	U
SW005	FI	BI00277EB	Selenium	AQAUTIC	4.1	MG/KG	U
SW005	FI	BI00277EB	Silver	AQAUTIC	2.1	MG/KG	U
SW005	FI	BI00277EB	Zinc	AQAUTIC	113	MG/KG	
SW026	FI	BI00475EB	Cadmium	AQAUTIC	1.4	MG/KG	U
SW026	FI	BI00475EB	Chromium	AQAUTIC	7.2	MG/KG	U
SW026	FI	BI00475EB	Copper	AQAUTIC	16.5	MG/KG	
SW026	FI	BI00475EB	Lead	AQAUTIC	3.0	MG/KG	
SW026	FI	BI00475EB	Mercury	AQAUTIC	0.7	MG/KG	U
SW026	FI	BI00475EB	Selenium	AQAUTIC	7.0	MG/KG	
SW026	FI	BI00475EB	Silver	AQAUTIC	4.3	MG/KG	U
SW026	FI	BI00475EB	Zinc	AQAUTIC	90.0	MG/KG	
SW026	FI	BI00476EB	Cadmium	AQAUTIC	2.5	MG/KG	U
SW026	FI	BI00476EB	Chromium	AQAUTIC	12.6	MG/KG	U
SW026	FI	BI00476EB	Copper	AQAUTIC	12.4	MG/KG	
SW026	FI	BI00476EB	Lead	AQAUTIC	3.2	MG/KG	
SW026	FI	BI00476EB	Mercury	AQAUTIC	1.3	MG/KG	U
SW026	FI	BI00476EB	Selenium	AQAUTIC	10.1	MG/KG	UI
SW026	FI	BI00476EB	Silver	AQAUTIC	7.5	MG/KG	U
SW026	FI	BI00476EB	Zinc	AQAUTIC	300	MG/KG	
SW026	FI	BI00477EB	Cadmium	AQAUTIC	2.4	MG/KG	U
SW026	FI	BI00477EB	Chromium	AQAUTIC	12.1	MG/KG	U
SW026	FI	BI00477EB	Copper	AQAUTIC	11.9	MG/KG	
SW026	FI	BI00477EB	Lead	AQAUTIC	5.8	MG/KG	
SW026	FI	BI00477EB	Mercury	AQAUTIC	0.9	MG/KG	U
SW026	FI	BI00477EB	Selenium	AQAUTIC	9.7	MG/KG	U
SW026	FI	BI00477EB	Silver	AQAUTIC	7.3	MG/KG	U
SW026	FI	BI00477EB	Zinc	AQAUTIC	193	MG/KG	
SW026	FI	BI00478EB	Cadmium	AQAUTIC	2.5	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SW026	FI	BI00478EB	Chromium	AQAUTIC	12.7	MG/KG	U
SW026	FI	BI00478EB	Copper	AQAUTIC	14.5	MG/KG	
SW026	FI	BI00478EB	Lead	AQAUTIC	3.6	MG/KG	
SW026	FI	BI00478EB	Mercury	AQAUTIC	1.3	MG/KG	U
SW026	FI	BI00478EB	Selenium	AQAUTIC	10.1	MG/KG	UI
SW026	FI	BI00478EB	Silver	AQAUTIC	7.6	MG/KG	U
SW026	FI	BI00478EB	Zinc	AQAUTIC	129	MG/KG	
SW026	FI	BI00479EB	Cadmium	AQAUTIC	2.5	MG/KG	U
SW026	FI	BI00479EB	Chromium	AQAUTIC	12.6	MG/KG	U
SW026	FI	BI00479EB	Copper	AQAUTIC	10.1	MG/KG	U
SW026	FI	BI00479EB	Lead	AQAUTIC	3.9	MG/KG	
SW026	FI	BI00479EB	Mercury	AQAUTIC	1.0	MG/KG	U
SW026	FI	BI00479EB	Selenium	AQAUTIC	10.1	MG/KG	UI
SW026	FI	BI00479EB	Silver	AQAUTIC	7.5	MG/KG	U
SW026	FI	BI00479EB	Zinc	AQAUTIC	272	MG/KG	
SW033	FI	BI00484EB	Cadmium	AQAUTIC	2.6	MG/KG	U
SW033	FI	BI00484EB	Chromium	AQAUTIC	13.2	MG/KG	U
SW033	FI	BI00484EB	Copper	AQAUTIC	10.5	MG/KG	U
SW033	FI	BI00484EB	Lead	AQAUTIC	5.7	MG/KG	
SW033	FI	BI00484EB	Mercury	AQAUTIC	1.3	MG/KG	U
SW033	FI	BI00484EB	Selenium	AQAUTIC	10.5	MG/KG	UI
SW033	FI	BI00484EB	Silver	AQAUTIC	7.9	MG/KG	U
SW033	FI	BI00484EB	Zinc	AQAUTIC	302	MG/KG	
SW033	FI	BI00485EB	Cadmium	AQAUTIC	2.6	MG/KG	U
SW033	FI	BI00485EB	Chromium	AQAUTIC	13.2	MG/KG	U
SW033	FI	BI00485EB	Copper	AQAUTIC	19.4	MG/KG	
SW033	FI	BI00485EB	Lead	AQAUTIC	3.4	MG/KG	
SW033	FI	BI00485EB	Mercury	AQAUTIC	1.3	MG/KG	U
SW033	FI	BI00485EB	Selenium	AQAUTIC	10.5	MG/KG	U
SW033	FI	BI00485EB	Silver	AQAUTIC	7.9	MG/KG	U
SW033	FI	BI00485EB	Zinc	AQAUTIC	296	MG/KG	
SW033	FI	BI00486EB	Cadmium	AQAUTIC	2.3	MG/KG	U
SW033	FI	BI00486EB	Chromium	AQAUTIC	11.5	MG/KG	U
SW033	FI	BI00486EB	Copper	AQAUTIC	13.5	MG/KG	
SW033	FI	BI00486EB	Lead	AQAUTIC	4.0	MG/KG	
SW033	FI	BI00486EB	Mercury	AQAUTIC	1.0	MG/KG	U
SW033	FI	BI00486EB	Selenium	AQAUTIC	9.2	MG/KG	U
SW033	FI	BI00486EB	Silver	AQAUTIC	6.9	MG/KG	U
SW033	FI	BI00486EB	Zinc	AQAUTIC	265	MG/KG	
SW038	BM	BI00501EB	Cadmium	AQAUTIC	2.7	MG/KG	U
SW038	BM	BI00501EB	Chromium	AQAUTIC	5.6	MG/KG	
SW038	BM	BI00501EB	Copper	AQAUTIC	122	MG/KG	
SW038	BM	BI00501EB	Lead	AQAUTIC	5.2	MG/KG	
SW038	BM	BI00501EB	Mercury	AQAUTIC	0.6	MG/KG	U
SW038	BM	BI00501EB	Selenium	AQAUTIC	5.5	MG/KG	UI
SW038	BM	BI00501EB	Silver	AQAUTIC	2.7	MG/KG	U
SW038	BM	BI00501EB	Zinc	AQAUTIC	219	MG/KG	
SWC001	BM	BI00211EB	Cadmium	AQAUTIC	5.2	MG/KG	U
SWC001	BM	BI00211EB	Chromium	AQAUTIC	7.7	MG/KG	U
SWC001	BM	BI00211EB	Copper	AQAUTIC	99.7	MG/KG	
SWC001	BM	BI00211EB	Lead	AQAUTIC	10.1	MG/KG	
SWC001	BM	BI00211EB	Mercury	AQAUTIC	1.1	MG/KG	U
SWC001	BM	BI00211EB	Selenium	AQAUTIC	10.3	MG/KG	U
SWC001	BM	BI00211EB	Silver	AQAUTIC	5.2	MG/KG	U
SWC001	BM	BI00211EB	Zinc	AQAUTIC	301	MG/KG	

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SWC001	BM	BI00212EB	Cadmium	AQAUTIC	5.2	MG/KG	U
SWC001	BM	BI00212EB	Chromium	AQAUTIC	15.1	MG/KG	
SWC001	BM	BI00212EB	Copper	AQAUTIC	604	MG/KG	
SWC001	BM	BI00212EB	Lead	AQAUTIC	12.7	MG/KG	
SWC001	BM	BI00212EB	Mercury	AQAUTIC	1.2	MG/KG	U
SWC001	BM	BI00212EB	Silver	AQAUTIC	5.2	MG/KG	U
SWC001	BM	BI00212EB	Zinc	AQAUTIC	406	MG/KG	
SWC001	BM	BI00500EB	Cadmium	AQAUTIC	3.0	MG/KG	U
SWC001	BM	BI00500EB	Chromium	AQAUTIC	4.7	MG/KG	
SWC001	BM	BI00500EB	Copper	AQAUTIC	118	MG/KG	
SWC001	BM	BI00500EB	Lead	AQAUTIC	4.1	MG/KG	
SWC001	BM	BI00500EB	Mercury	AQAUTIC	0.7	MG/KG	U
SWC001	BM	BI00500EB	Selenium	AQAUTIC	6.1	MG/KG	UI
SWC001	BM	BI00500EB	Silver	AQAUTIC	3.0	MG/KG	U
SWC001	BM	BI00500EB	Zinc	AQAUTIC	126	MG/KG	
SWC001	FI	BI00203EB	Cadmium	AQAUTIC	5.0	MG/KG	U
SWC001	FI	BI00203EB	Chromium	AQAUTIC	8.9	MG/KG	
SWC001	FI	BI00203EB	Copper	AQAUTIC	73.0	MG/KG	
SWC001	FI	BI00203EB	Lead	AQAUTIC	4.3	MG/KG	
SWC001	FI	BI00203EB	Mercury	AQAUTIC	1.3	MG/KG	U
SWC001	FI	BI00203EB	Selenium	AQAUTIC	10.1	MG/KG	UI
SWC001	FI	BI00203EB	Silver	AQAUTIC	5.0	MG/KG	U
SWC001	FI	BI00203EB	Zinc	AQAUTIC	426	MG/KG	
SWC001	FI	BI00204EB	Cadmium	AQAUTIC	1.3	MG/KG	U
SWC001	FI	BI00204EB	Chromium	AQAUTIC	2.4	MG/KG	
SWC001	FI	BI00204EB	Copper	AQAUTIC	1.7	MG/KG	
SWC001	FI	BI00204EB	Lead	AQAUTIC	0.9	MG/KG	
SWC001	FI	BI00204EB	Mercury	AQAUTIC	0.3	MG/KG	
SWC001	FI	BI00204EB	Selenium	AQAUTIC	2.7	MG/KG	U
SWC001	FI	BI00204EB	Silver	AQAUTIC	1.3	MG/KG	U
SWC001	FI	BI00204EB	Zinc	AQAUTIC	96.1	MG/KG	
SWC001	FI	BI00205EB	Cadmium	AQAUTIC	3.1	MG/KG	U
SWC001	FI	BI00205EB	Chromium	AQAUTIC	4.6	MG/KG	U
SWC001	FI	BI00205EB	Copper	AQAUTIC	52.6	MG/KG	
SWC001	FI	BI00205EB	Lead	AQAUTIC	2.6	MG/KG	
SWC001	FI	BI00205EB	Mercury	AQAUTIC	0.6	MG/KG	U
SWC001	FI	BI00205EB	Selenium	AQAUTIC	6.1	MG/KG	UI
SWC001	FI	BI00205EB	Silver	AQAUTIC	3.1	MG/KG	U
SWC001	FI	BI00205EB	Zinc	AQAUTIC	104	MG/KG	
SWC001	FI	BI00206EB	Cadmium	AQAUTIC	2.4	MG/KG	U
SWC001	FI	BI00206EB	Chromium	AQAUTIC	3.6	MG/KG	U
SWC001	FI	BI00206EB	Copper	AQAUTIC	9.9	MG/KG	
SWC001	FI	BI00206EB	Lead	AQAUTIC	1.7	MG/KG	
SWC001	FI	BI00206EB	Mercury	AQAUTIC	0.6	MG/KG	U
SWC001	FI	BI00206EB	Selenium	AQAUTIC	4.8	MG/KG	U
SWC001	FI	BI00206EB	Silver	AQAUTIC	2.4	MG/KG	U
SWC001	FI	BI00206EB	Zinc	AQAUTIC	73.2	MG/KG	
SWC001	FI	BI00210EB	Cadmium	AQAUTIC	8.3	MG/KG	U
SWC001	FI	BI00210EB	Chromium	AQAUTIC	12.5	MG/KG	U
SWC001	FI	BI00210EB	Copper	AQAUTIC	657	MG/KG	
SWC001	FI	BI00210EB	Lead	AQAUTIC	6.4	MG/KG	
SWC001	FI	BI00210EB	Mercury	AQAUTIC	1.7	MG/KG	U
SWC001	FI	BI00210EB	Silver	AQAUTIC	8.3	MG/KG	U
SWC001	FI	BI00210EB	Zinc	AQAUTIC	539	MG/KG	
SWC001	FI	BI00213EB	Cadmium	AQAUTIC	3.2	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SWC001	FI	BI00213EB	Chromium	AQAUTIC	10.4	MG/KG	
SWC001	FI	BI00213EB	Copper	AQAUTIC	103	MG/KG	
SWC001	FI	BI00213EB	Lead	AQAUTIC	3.0	MG/KG	
SWC001	FI	BI00213EB	Mercury	AQAUTIC	0.8	MG/KG	U
SWC001	FI	BI00213EB	Selenium	AQAUTIC	6.3	MG/KG	U
SWC001	FI	BI00213EB	Silver	AQAUTIC	3.2	MG/KG	U
SWC001	FI	BI00213EB	Zinc	AQAUTIC	158	MG/KG	
SWC001	FI	BI00215EB	Cadmium	AQAUTIC	1.4	MG/KG	U
SWC001	FI	BI00215EB	Chromium	AQAUTIC	2.1	MG/KG	U
SWC001	FI	BI00215EB	Copper	AQAUTIC	1.4	MG/KG	U
SWC001	FI	BI00215EB	Lead	AQAUTIC	1.4	MG/KG	
SWC001	FI	BI00215EB	Mercury	AQAUTIC	0.3	MG/KG	U
SWC001	FI	BI00215EB	Selenium	AQAUTIC	3.0	MG/KG	
SWC001	FI	BI00215EB	Silver	AQAUTIC	1.4	MG/KG	U
SWC001	FI	BI00215EB	Zinc	AQAUTIC	40.5	MG/KG	
SWC001	FI	BI00216EB	Cadmium	AQAUTIC	1.3	MG/KG	U
SWC001	FI	BI00216EB	Chromium	AQAUTIC	2.4	MG/KG	
SWC001	FI	BI00216EB	Copper	AQAUTIC	5.9	MG/KG	
SWC001	FI	BI00216EB	Lead	AQAUTIC	1.0	MG/KG	
SWC001	FI	BI00216EB	Mercury	AQAUTIC	0.3	MG/KG	U
SWC001	FI	BI00216EB	Selenium	AQAUTIC	2.7	MG/KG	UI
SWC001	FI	BI00216EB	Silver	AQAUTIC	1.3	MG/KG	U
SWC001	FI	BI00216EB	Zinc	AQAUTIC	44.5	MG/KG	
SWC001	FI	BI00217EB	Cadmium	AQAUTIC	1.4	MG/KG	U
SWC001	FI	BI00217EB	Chromium	AQAUTIC	3.1	MG/KG	
SWC001	FI	BI00217EB	Copper	AQAUTIC	3.5	MG/KG	
SWC001	FI	BI00217EB	Lead	AQAUTIC	1.7	MG/KG	
SWC001	FI	BI00217EB	Mercury	AQAUTIC	0.3	MG/KG	U
SWC001	FI	BI00217EB	Selenium	AQAUTIC	21.3	MG/KG	
SWC001	FI	BI00217EB	Silver	AQAUTIC	1.4	MG/KG	U
SWC001	FI	BI00217EB	Zinc	AQAUTIC	62.1	MG/KG	
SWC001	FI	BI00218EB	Cadmium	AQAUTIC	1.3	MG/KG	
SWC001	FI	BI00218EB	Cadmium	AQAUTIC	1.3	MG/KG	
SWC001	FI	BI00218EB	Chromium	AQAUTIC	4.0	MG/KG	U
SWC001	FI	BI00218EB	Chromium	AQAUTIC	4.5	MG/KG	
SWC001	FI	BI00218EB	Copper	AQAUTIC	9.4	MG/KG	
SWC001	FI	BI00218EB	Copper	AQAUTIC	9.8	MG/KG	
SWC001	FI	BI00218EB	Lead	AQAUTIC	1.2	MG/KG	
SWC001	FI	BI00218EB	Lead	AQAUTIC	0.9	MG/KG	
SWC001	FI	BI00218EB	Mercury	AQAUTIC	0.4	MG/KG	
SWC001	FI	BI00218EB	Mercury	AQAUTIC	0.4	MG/KG	
SWC001	FI	BI00218EB	Selenium	AQAUTIC	3.2	MG/KG	U
SWC001	FI	BI00218EB	Selenium	AQAUTIC	3.2	MG/KG	U
SWC001	FI	BI00218EB	Silver	AQAUTIC	2.4	MG/KG	U
SWC001	FI	BI00218EB	Silver	AQAUTIC	2.4	MG/KG	U
SWC001	FI	BI00218EB	Zinc	AQAUTIC	60.1	MG/KG	
SWC001	FI	BI00218EB	Zinc	AQAUTIC	63.6	MG/KG	
SWC001	FI	BI00219EB	Cadmium	AQAUTIC	97.0	%	
SWC001	FI	BI00219EB	Cadmium	AQAUTIC	1.3	MG/KG	
SWC001	FI	BI00219EB	Chromium	AQAUTIC	3.7	MG/KG	U
SWC001	FI	BI00219EB	Chromium	AQAUTIC	95.3	%	U
SWC001	FI	BI00219EB	Copper	AQAUTIC	97.0	%	
SWC001	FI	BI00219EB	Copper	AQAUTIC	8.1	MG/KG	
SWC001	FI	BI00219EB	Lead	AQAUTIC	0.7	MG/KG	U
SWC001	FI	BI00219EB	Lead	AQAUTIC	95.9	%	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SWC001	FI	BI00219EB	Mercury	AQAUTIC	114	%	U
SWC001	FI	BI00219EB	Mercury	AQAUTIC	0.3	MG/KG	U
SWC001	FI	BI00219EB	Selenium	AQAUTIC	3.0	MG/KG	UI
SWC001	FI	BI00219EB	Selenium	AQAUTIC	130	%	U
SWC001	FI	BI00219EB	Silver	AQAUTIC	87.8	%	U
SWC001	FI	BI00219EB	Silver	AQAUTIC	2.2	MG/KG	U
SWC001	FI	BI00219EB	Zinc	AQAUTIC	73.8	MG/KG	
SWC001	FI	BI00219EB	Zinc	AQAUTIC	89.3	%	
SWC001	FI	BI00220EB	Cadmium	AQAUTIC	3.3	MG/KG	
SWC001	FI	BI00220EB	Chromium	AQAUTIC	10.2	MG/KG	U
SWC001	FI	BI00220EB	Copper	AQAUTIC	18.9	MG/KG	
SWC001	FI	BI00220EB	Lead	AQAUTIC	2.3	MG/KG	
SWC001	FI	BI00220EB	Mercury	AQAUTIC	0.8	MG/KG	U
SWC001	FI	BI00220EB	Selenium	AQAUTIC	8.2	MG/KG	U
SWC001	FI	BI00220EB	Silver	AQAUTIC	6.1	MG/KG	U
SWC001	FI	BI00220EB	Zinc	AQAUTIC	103	MG/KG	
SWC001	FI	BI00221EB	Cadmium	AQAUTIC	3.9	MG/KG	
SWC001	FI	BI00221EB	Chromium	AQAUTIC	11.7	MG/KG	U
SWC001	FI	BI00221EB	Copper	AQAUTIC	18.0	MG/KG	
SWC001	FI	BI00221EB	Lead	AQAUTIC	2.3	MG/KG	UI
SWC001	FI	BI00221EB	Mercury	AQAUTIC	1.1	MG/KG	U
SWC001	FI	BI00221EB	Selenium	AQAUTIC	9.3	MG/KG	U
SWC001	FI	BI00221EB	Silver	AQAUTIC	7.0	MG/KG	U
SWC001	FI	BI00221EB	Zinc	AQAUTIC	187	MG/KG	
SWC002	AV	BI00525EB	Cadmium	AQAUTIC	3.7	MG/KG	
SWC002	AV	BI00525EB	Chromium	AQAUTIC	4.5	MG/KG	U
SWC002	AV	BI00525EB	Copper	AQAUTIC	13.8	MG/KG	
SWC002	AV	BI00525EB	Lead	AQAUTIC	1.5	MG/KG	UI
SWC002	AV	BI00525EB	Mercury	AQAUTIC	0.7	MG/KG	U
SWC002	AV	BI00525EB	Selenium	AQAUTIC	6.0	MG/KG	UI
SWC002	AV	BI00525EB	Silver	AQAUTIC	3.0	MG/KG	U
SWC002	AV	BI00525EB	Zinc	AQAUTIC	89.9	MG/KG	
SWC002	FI	BI00472EB	Cadmium	AQAUTIC	1.3	MG/KG	
SWC002	FI	BI00472EB	Cadmium	AQAUTIC	1.1	MG/KG	U
SWC002	FI	BI00472EB	Chromium	AQAUTIC	5.7	MG/KG	U
SWC002	FI	BI00472EB	Chromium	AQAUTIC	5.7	MG/KG	U
SWC002	FI	BI00472EB	Copper	AQAUTIC	12.3	MG/KG	
SWC002	FI	BI00472EB	Copper	AQAUTIC	10.4	MG/KG	
SWC002	FI	BI00472EB	Lead	AQAUTIC	2.4	MG/KG	
SWC002	FI	BI00472EB	Lead	AQAUTIC	4.7	MG/KG	
SWC002	FI	BI00472EB	Mercury	AQAUTIC	0.5	MG/KG	U
SWC002	FI	BI00472EB	Mercury	AQAUTIC	0.5	MG/KG	U
SWC002	FI	BI00472EB	Selenium	AQAUTIC	4.9	MG/KG	
SWC002	FI	BI00472EB	Selenium	AQAUTIC	4.6	MG/KG	U
SWC002	FI	BI00472EB	Silver	AQAUTIC	3.4	MG/KG	U
SWC002	FI	BI00472EB	Silver	AQAUTIC	3.4	MG/KG	U
SWC002	FI	BI00472EB	Zinc	AQAUTIC	399	MG/KG	
SWC002	FI	BI00472EB	Zinc	AQAUTIC	281	MG/KG	
SWC002	FI	BI00473EB	Cadmium	AQAUTIC	1.3	MG/KG	U
SWC002	FI	BI00473EB	Cadmium	AQAUTIC	95.2	%	U
SWC002	FI	BI00473EB	Chromium	AQAUTIC	95.2	%	U
SWC002	FI	BI00473EB	Chromium	AQAUTIC	6.5	MG/KG	U
SWC002	FI	BI00473EB	Copper	AQAUTIC	54.6	MG/KG	
SWC002	FI	BI00473EB	Copper	AQAUTIC	98.4	%	
SWC002	FI	BI00473EB	Lead	AQAUTIC	97.2	%	

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SWC002	FI	BI00473EB	Lead	AQAUTIC	3.1	MG/KG	
SWC002	FI	BI00473EB	Mercury	AQAUTIC	0.6	MG/KG	U
SWC002	FI	BI00473EB	Mercury	AQAUTIC	79.0	%	U
SWC002	FI	BI00473EB	Selenium	AQAUTIC	53.9	%	U
SWC002	FI	BI00473EB	Selenium	AQAUTIC	5.2	MG/KG	U
SWC002	FI	BI00473EB	Silver	AQAUTIC	3.9	MG/KG	U
SWC002	FI	BI00473EB	Silver	AQAUTIC	84.6	%	U
SWC002	FI	BI00473EB	Zinc	AQAUTIC	433	MG/KG	
SWC002	FI	BI00473EB	Zinc	AQAUTIC	86.1	%	
SWC002	FI	BI00474EB	Cadmium	AQAUTIC	1.4	MG/KG	U
SWC002	FI	BI00474EB	Chromium	AQAUTIC	7.1	MG/KG	U
SWC002	FI	BI00474EB	Copper	AQAUTIC	8.4	MG/KG	
SWC002	FI	BI00474EB	Lead	AQAUTIC	3.2	MG/KG	
SWC002	FI	BI00474EB	Mercury	AQAUTIC	0.6	MG/KG	U
SWC002	FI	BI00474EB	Selenium	AQAUTIC	5.7	MG/KG	UI
SWC002	FI	BI00474EB	Silver	AQAUTIC	4.3	MG/KG	U
SWC002	FI	BI00474EB	Zinc	AQAUTIC	538	MG/KG	
SWC002	HE	BI00233EB	Cadmium	AQAUTIC	1.3	MG/KG	U
SWC002	HE	BI00233EB	Chromium	AQAUTIC	6.6	MG/KG	U
SWC002	HE	BI00233EB	Copper	AQAUTIC	16.0	MG/KG	
SWC002	HE	BI00233EB	Lead	AQAUTIC	2.0	MG/KG	
SWC002	HE	BI00233EB	Mercury	AQAUTIC	0.5	MG/KG	U
SWC002	HE	BI00233EB	Selenium	AQAUTIC	5.3	MG/KG	UI
SWC002	HE	BI00233EB	Silver	AQAUTIC	4.0	MG/KG	U
SWC002	HE	BI00233EB	Zinc	AQAUTIC	53.4	MG/KG	
SWC002	HE	BI00234EB	Cadmium	AQAUTIC	1.5	MG/KG	U
SWC002	HE	BI00234EB	Chromium	AQAUTIC	7.5	MG/KG	U
SWC002	HE	BI00234EB	Copper	AQAUTIC	6.0	MG/KG	U
SWC002	HE	BI00234EB	Lead	AQAUTIC	2.8	MG/KG	
SWC002	HE	BI00234EB	Mercury	AQAUTIC	0.6	MG/KG	U
SWC002	HE	BI00234EB	Selenium	AQAUTIC	6.0	MG/KG	U
SWC002	HE	BI00234EB	Silver	AQAUTIC	4.5	MG/KG	U
SWC002	HE	BI00234EB	Zinc	AQAUTIC	33.8	MG/KG	
SWC002	HE	BI00235EB	Cadmium	AQAUTIC	1.0	MG/KG	U
SWC002	HE	BI00235EB	Chromium	AQAUTIC	5.2	MG/KG	U
SWC002	HE	BI00235EB	Copper	AQAUTIC	4.1	MG/KG	U
SWC002	HE	BI00235EB	Lead	AQAUTIC	1.6	MG/KG	I
SWC002	HE	BI00235EB	Mercury	AQAUTIC	0.4	MG/KG	U
SWC002	HE	BI00235EB	Selenium	AQAUTIC	4.1	MG/KG	UI
SWC002	HE	BI00235EB	Silver	AQAUTIC	3.1	MG/KG	U
SWC002	HE	BI00235EB	Zinc	AQAUTIC	25.2	MG/KG	
SWC002	HE	BI00237EB	Cadmium	AQAUTIC	3.3	MG/KG	
SWC002	HE	BI00237EB	Chromium	AQAUTIC	10.0	MG/KG	U
SWC002	HE	BI00237EB	Copper	AQAUTIC	57.2	MG/KG	
SWC002	HE	BI00237EB	Lead	AQAUTIC	5.7	MG/KG	
SWC002	HE	BI00237EB	Mercury	AQAUTIC	0.9	MG/KG	U
SWC002	HE	BI00237EB	Selenium	AQAUTIC	8.0	MG/KG	U
SWC002	HE	BI00237EB	Silver	AQAUTIC	6.0	MG/KG	U
SWC002	HE	BI00237EB	Zinc	AQAUTIC	233	MG/KG	
WOP002	BM	BI00496EB	Cadmium	AQAUTIC	3.1	MG/KG	U
WOP002	BM	BI00496EB	Chromium	AQAUTIC	4.9	MG/KG	
WOP002	BM	BI00496EB	Copper	AQAUTIC	516	MG/KG	
WOP002	BM	BI00496EB	Lead	AQAUTIC	5.6	MG/KG	
WOP002	BM	BI00496EB	Mercury	AQAUTIC	0.6	MG/KG	U
WOP002	BM	BI00496EB	Selenium	AQAUTIC	6.1	MG/KG	UI



## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
WOPO02	BM	BI00496EB	Silver	AQAUTIC	3.1	MG/KG	U
WOPO02	BM	BI00496EB	Zinc	AQAUTIC	203	MG/KG	
WOPO02	BM	BI00497EB	Cadmium	AQAUTIC	2.6	MG/KG	U
WOPO02	BM	BI00497EB	Chromium	AQAUTIC	5.8	MG/KG	
WOPO02	BM	BI00497EB	Copper	AQAUTIC	135	MG/KG	
WOPO02	BM	BI00497EB	Lead	AQAUTIC	4.0	MG/KG	
WOPO02	BM	BI00497EB	Mercury	AQAUTIC	0.6	MG/KG	U
WOPO02	BM	BI00497EB	Selenium	AQAUTIC	5.1	MG/KG	UI
WOPO02	BM	BI00497EB	Silver	AQAUTIC	3.3	MG/KG	
WOPO02	BM	BI00497EB	Zinc	AQAUTIC	145	MG/KG	
WOPO02	FI	BI00480EB	Cadmium	AQAUTIC	3.4	MG/KG	U
WOPO02	FI	BI00480EB	Chromium	AQAUTIC	17.2	MG/KG	U
WOPO02	FI	BI00480EB	Copper	AQAUTIC	115	MG/KG	
WOPO02	FI	BI00480EB	Lead	AQAUTIC	4.4	MG/KG	
WOPO02	FI	BI00480EB	Mercury	AQAUTIC	1.7	MG/KG	U
WOPO02	FI	BI00480EB	Selenium	AQAUTIC	14.3	MG/KG	
WOPO02	FI	BI00480EB	Silver	AQAUTIC	10.3	MG/KG	U
WOPO02	FI	BI00480EB	Zinc	AQAUTIC	120	MG/KG	
WOPO02	FI	BI00481EB	Cadmium	AQAUTIC	1.9	MG/KG	U
WOPO02	FI	BI00481EB	Chromium	AQAUTIC	9.6	MG/KG	U
WOPO02	FI	BI00481EB	Copper	AQAUTIC	7.7	MG/KG	U
WOPO02	FI	BI00481EB	Lead	AQAUTIC	3.5	MG/KG	
WOPO02	FI	BI00481EB	Mercury	AQAUTIC	0.9	MG/KG	U
WOPO02	FI	BI00481EB	Selenium	AQAUTIC	7.7	MG/KG	UI
WOPO02	FI	BI00481EB	Silver	AQAUTIC	5.8	MG/KG	U
WOPO02	FI	BI00481EB	Zinc	AQAUTIC	185	MG/KG	
WOPO02	FI	BI00482EB	Cadmium	AQAUTIC	2.1	MG/KG	U
WOPO02	FI	BI00482EB	Chromium	AQAUTIC	10.6	MG/KG	U
WOPO02	FI	BI00482EB	Copper	AQAUTIC	8.5	MG/KG	U
WOPO02	FI	BI00482EB	Lead	AQAUTIC	2.4	MG/KG	
WOPO02	FI	BI00482EB	Mercury	AQAUTIC	0.9	MG/KG	U
WOPO02	FI	BI00482EB	Selenium	AQAUTIC	8.5	MG/KG	UI
WOPO02	FI	BI00482EB	Silver	AQAUTIC	6.4	MG/KG	U
WOPO02	FI	BI00482EB	Zinc	AQAUTIC	145	MG/KG	
WOPO02	FI	BI00483EB	Cadmium	AQAUTIC	1.8	MG/KG	U
WOPO02	FI	BI00483EB	Chromium	AQAUTIC	8.8	MG/KG	U
WOPO02	FI	BI00483EB	Copper	AQAUTIC	7.1	MG/KG	U
WOPO02	FI	BI00483EB	Lead	AQAUTIC	3.8	MG/KG	
WOPO02	FI	BI00483EB	Mercury	AQAUTIC	0.8	MG/KG	U
WOPO02	FI	BI00483EB	Selenium	AQAUTIC	7.1	MG/KG	U
WOPO02	FI	BI00483EB	Silver	AQAUTIC	5.3	MG/KG	U
WOPO02	FI	BI00483EB	Zinc	AQAUTIC	192	MG/KG	
WORI01	BM	BI00487EB	Cadmium	AQAUTIC	3.6	MG/KG	U
WORI01	BM	BI00487EB	Chromium	AQAUTIC	17.9	MG/KG	U
WORI01	BM	BI00487EB	Copper	AQAUTIC	804	MG/KG	
WORI01	BM	BI00487EB	Lead	AQAUTIC	11.4	MG/KG	
WORI01	BM	BI00487EB	Mercury	AQAUTIC	1.7	MG/KG	U
WORI01	BM	BI00487EB	Selenium	AQAUTIC	14.3	MG/KG	UI
WORI01	BM	BI00487EB	Silver	AQAUTIC	10.7	MG/KG	U
WORI01	BM	BI00487EB	Zinc	AQAUTIC	375	MG/KG	
WORI01	FI	BI00488EB	Cadmium	AQAUTIC	0.5	MG/KG	U
WORI01	FI	BI00488EB	Chromium	AQAUTIC	3.0	MG/KG	U
WORI01	FI	BI00488EB	Copper	AQAUTIC	21.7	MG/KG	
WORI01	FI	BI00488EB	Lead	AQAUTIC	2.1	MG/KG	
WORI01	FI	BI00488EB	Mercury	AQAUTIC	0.2	MG/KG	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
WOR101	FI	BI00488EB	Selenium	AQAUTIC	2.4	MG/KG	U
WOR101	FI	BI00488EB	Silver	AQAUTIC	1.8	MG/KG	U
WOR101	FI	BI00488EB	Zinc	AQAUTIC	141	MG/KG	
WOR101	FI	BI00489EB	Cadmium	AQAUTIC	1.1	MG/KG	U
WOR101	FI	BI00489EB	Chromium	AQAUTIC	5.7	MG/KG	U
WOR101	FI	BI00489EB	Copper	AQAUTIC	14.0	MG/KG	
WOR101	FI	BI00489EB	Lead	AQAUTIC	1.3	MG/KG	
WOR101	FI	BI00489EB	Mercury	AQAUTIC	0.5	MG/KG	U
WOR101	FI	BI00489EB	Selenium	AQAUTIC	4.6	MG/KG	UI
WOR101	FI	BI00489EB	Silver	AQAUTIC	3.4	MG/KG	U
WOR101	FI	BI00489EB	Zinc	AQAUTIC	126	MG/KG	
WOR101	FI	BI00490EB	Cadmium	AQAUTIC	0.9	MG/KG	U
WOR101	FI	BI00490EB	Chromium	AQAUTIC	4.9	MG/KG	U
WOR101	FI	BI00490EB	Copper	AQAUTIC	3.9	MG/KG	U
WOR101	FI	BI00490EB	Lead	AQAUTIC	1.5	MG/KG	
WOR101	FI	BI00490EB	Mercury	AQAUTIC	0.4	MG/KG	U
WOR101	FI	BI00490EB	Selenium	AQAUTIC	3.9	MG/KG	UI
WOR101	FI	BI00490EB	Silver	AQAUTIC	2.9	MG/KG	U
WOR101	FI	BI00490EB	Zinc	AQAUTIC	85.4	MG/KG	
WOR101	FI	BI00491EB	Cadmium	AQAUTIC	1.0	MG/KG	U
WOR101	FI	BI00491EB	Chromium	AQAUTIC	5.0	MG/KG	U
WOR101	FI	BI00491EB	Copper	AQAUTIC	4.1	MG/KG	
WOR101	FI	BI00491EB	Lead	AQAUTIC	1.7	MG/KG	
WOR101	FI	BI00491EB	Mercury	AQAUTIC	0.4	MG/KG	U
WOR101	FI	BI00491EB	Selenium	AQAUTIC	4.0	MG/KG	UI
WOR101	FI	BI00491EB	Silver	AQAUTIC	3.0	MG/KG	U
WOR101	FI	BI00491EB	Zinc	AQAUTIC	89.7	MG/KG	
WOR101	FI	BI00492EB	Cadmium	AQAUTIC	1.7	MG/KG	U
WOR101	FI	BI00492EB	Cadmium	AQAUTIC	1.7	MG/KG	U
WOR101	FI	BI00492EB	Chromium	AQAUTIC	3.9	MG/KG	
WOR101	FI	BI00492EB	Chromium	AQAUTIC	2.6	MG/KG	U
WOR101	FI	BI00492EB	Copper	AQAUTIC	76.9	MG/KG	
WOR101	FI	BI00492EB	Copper	AQAUTIC	83.9	MG/KG	
WOR101	FI	BI00492EB	Lead	AQAUTIC	0.9	MG/KG	
WOR101	FI	BI00492EB	Lead	AQAUTIC	2.8	MG/KG	
WOR101	FI	BI00492EB	Mercury	AQAUTIC	0.4	MG/KG	U
WOR101	FI	BI00492EB	Mercury	AQAUTIC	0.4	MG/KG	U
WOR101	FI	BI00492EB	Selenium	AQAUTIC	3.5	MG/KG	UI
WOR101	FI	BI00492EB	Selenium	AQAUTIC	3.5	MG/KG	U
WOR101	FI	BI00492EB	Silver	AQAUTIC	1.7	MG/KG	U
WOR101	FI	BI00492EB	Silver	AQAUTIC	1.7	MG/KG	U
WOR101	FI	BI00492EB	Zinc	AQAUTIC	155	MG/KG	
WOR101	FI	BI00492EB	Zinc	AQAUTIC	158	MG/KG	
WOR101	FI	BI00493EB	Cadmium	AQAUTIC	1.9	MG/KG	U
WOR101	FI	BI00493EB	Cadmium	AQAUTIC	84.5	%	U
WOR101	FI	BI00493EB	Chromium	AQAUTIC	2.8	MG/KG	U
WOR101	FI	BI00493EB	Chromium	AQAUTIC	90.9	%	U
WOR101	FI	BI00493EB	Copper	AQAUTIC	34.2	MG/KG	
WOR101	FI	BI00493EB	Copper	AQAUTIC	80.9	%	
WOR101	FI	BI00493EB	Lead	AQAUTIC	0.9	MG/KG	
WOR101	FI	BI00493EB	Lead	AQAUTIC	107	%	
WOR101	FI	BI00493EB	Mercury	AQAUTIC	0.4	MG/KG	U
WOR101	FI	BI00493EB	Mercury	AQAUTIC	84.0	%	U
WOR101	FI	BI00493EB	Selenium	AQAUTIC	3.7	MG/KG	UI
WOR101	FI	BI00493EB	Selenium	AQAUTIC	63.5	%	U

## METAL RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
WORIO1	FI	BI00493EB	Silver	AQAUTIC	1.9	MG/KG	U
WORIO1	FI	BI00493EB	Silver	AQAUTIC	86.3	%	U
WORIO1	FI	BI00493EB	Zinc	AQAUTIC	205	MG/KG	
WORIO1	FI	BI00493EB	Zinc	AQAUTIC	94.0	%	
WORIO1	FI	BI00494EB	Cadmium	AQAUTIC	1.6	MG/KG	U
WORIO1	FI	BI00494EB	Chromium	AQAUTIC	2.5	MG/KG	
WORIO1	FI	BI00494EB	Copper	AQAUTIC	17.7	MG/KG	
WORIO1	FI	BI00494EB	Lead	AQAUTIC	0.9	MG/KG	
WORIO1	FI	BI00494EB	Mercury	AQAUTIC	0.3	MG/KG	U
WORIO1	FI	BI00494EB	Selenium	AQAUTIC	3.2	MG/KG	UI
WORIO1	FI	BI00494EB	Silver	AQAUTIC	1.6	MG/KG	U
WORIO1	FI	BI00494EB	Zinc	AQAUTIC	223	MG/KG	
WORIO3	FI	BI00495EB	Cadmium	AQAUTIC	2.0	MG/KG	U
WORIO3	FI	BI00495EB	Chromium	AQAUTIC	3.0	MG/KG	U
WORIO3	FI	BI00495EB	Copper	AQAUTIC	58.4	MG/KG	
WORIO3	FI	BI00495EB	Lead	AQAUTIC	1.0	MG/KG	U
WORIO3	FI	BI00495EB	Mercury	AQAUTIC	0.5	MG/KG	U
WORIO3	FI	BI00495EB	Selenium	AQAUTIC	4.0	MG/KG	U
WORIO3	FI	BI00495EB	Silver	AQAUTIC	2.0	MG/KG	U
WORIO3	FI	BI00495EB	Zinc	AQAUTIC	55.9	MG/KG	
WORIO3	FI	BI00498EB	Cadmium	AQAUTIC	1.7	MG/KG	U
WORIO3	FI	BI00498EB	Chromium	AQAUTIC	2.6	MG/KG	U
WORIO3	FI	BI00498EB	Copper	AQAUTIC	65.7	MG/KG	
WORIO3	FI	BI00498EB	Lead	AQAUTIC	0.9	MG/KG	
WORIO3	FI	BI00498EB	Mercury	AQAUTIC	0.4	MG/KG	U
WORIO3	FI	BI00498EB	Selenium	AQAUTIC	3.5	MG/KG	UI
WORIO3	FI	BI00498EB	Silver	AQAUTIC	1.7	MG/KG	U
WORIO3	FI	BI00498EB	Zinc	AQAUTIC	106	MG/KG	
WORIO3	FI	BI00499EB	Cadmium	AQAUTIC	1.9	MG/KG	U
WORIO3	FI	BI00499EB	Chromium	AQAUTIC	2.9	MG/KG	U
WORIO3	FI	BI00499EB	Copper	AQAUTIC	13.8	MG/KG	
WORIO3	FI	BI00499EB	Lead	AQAUTIC	0.9	MG/KG	U
WORIO3	FI	BI00499EB	Mercury	AQAUTIC	0.3	MG/KG	U
WORIO3	FI	BI00499EB	Selenium	AQAUTIC	3.8	MG/KG	UI
WORIO3	FI	BI00499EB	Silver	AQAUTIC	1.9	MG/KG	U
WORIO3	FI	BI00499EB	Zinc	AQAUTIC	56.9	MG/KG	
WOSP01	HE	BI00502EB	Cadmium	AQAUTIC	1.5	MG/KG	U
WOSP01	HE	BI00502EB	Chromium	AQAUTIC	7.5	MG/KG	U
WOSP01	HE	BI00502EB	Copper	AQAUTIC	48.9	MG/KG	
WOSP01	HE	BI00502EB	Lead	AQAUTIC	2.6	MG/KG	
WOSP01	HE	BI00502EB	Mercury	AQAUTIC	0.7	MG/KG	U
WOSP01	HE	BI00502EB	Selenium	AQAUTIC	9.2	MG/KG	
WOSP01	HE	BI00502EB	Silver	AQAUTIC	4.5	MG/KG	U
WOSP01	HE	BI00502EB	Zinc	AQAUTIC	58.2	MG/KG	
		BI00444EB	Cadmium		93.7	%	U
		BI00444EB	Chromium		94.4	%	U
		BI00444EB	Copper		98.1	%	
		BI00444EB	Lead		93.2	%	
		BI00444EB	Mercury		94.0	%	U
		BI00444EB	Selenium		33.8	%	U
		BI00444EB	Silver		62.8	%	U
		BI00444EB	Zinc		89.9	%	

1262 rows selected.

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MA01A	SM	BI00191EB	Americium 241	HYDRIC	0.001	pCi/g	J
MA01A	SM	BI00191EB	Plutonium 238	HYDRIC	0.001	pCi/g	U
MA01A	SM	BI00191EB	Plutonium 239/240	HYDRIC	0.002	pCi/g	J
MA01A	SM	BI00191EB	Radium 226	HYDRIC		pCi/g	
MA01A	SM	BI00191EB	Total Uranium	HYDRIC		pCi/g	
MA01A	VE	BI00289EB	Americium 241	HYDRIC	0	pCi/g	U
MA01A	VE	BI00289EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA01A	VE	BI00289EB	Plutonium 239/240	HYDRIC	0	pCi/g	U
MA01A	VE	BI00289EB	Radium 226	HYDRIC		pCi/g	
MA01A	VE	BI00289EB	Total Uranium	HYDRIC		pCi/g	
MA01A	VE	BI00290EB	Americium 241	HYDRIC	0	pCi/g	U
MA01A	VE	BI00290EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA01A	VE	BI00290EB	Plutonium 239/240	HYDRIC	0	pCi/g	U
MA01A	VE	BI00290EB	Radium 226	HYDRIC		pCi/g	
MA01A	VE	BI00290EB	Total Uranium	HYDRIC		pCi/g	
MA01A/3A	SM	BI00524EB	Americium 241	HYDRIC	0	pCi/g	U
MA01A/3A	SM	BI00524EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA01A/3A	SM	BI00524EB	Plutonium 239/240	HYDRIC	0.002	pCi/g	J
MA01A/3A	SM	BI00524EB	Radium 226	HYDRIC		pCi/g	
MA01A/3A	SM	BI00524EB	Total Uranium	HYDRIC		pCi/g	
MA01R	SM	BI00247EB	Americium 241	HYDRIC	0.002	pCi/g	U
MA01R	SM	BI00247EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA01R	SM	BI00247EB	Plutonium 239/240	HYDRIC	0	pCi/g	U
MA01R	SM	BI00247EB	Radium 226	HYDRIC		pCi/g	
MA01R	SM	BI00247EB	Total Uranium	HYDRIC	0.17	pCi/g	J
MA01R	TA	BI00328EB	Americium 241	HYDRIC	0.002	pCi/g	U
MA01R	TA	BI00328EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA01R	TA	BI00328EB	Plutonium 239/240	HYDRIC	0.015	pCi/g	BJ
MA01R	TA	BI00328EB	Radium 226	HYDRIC		pCi/g	
MA01R	TA	BI00328EB	Total Uranium	HYDRIC		pCi/g	
MA01R	VE	BI00329EB	Americium 241	HYDRIC	0.001	pCi/g	U
MA01R	VE	BI00329EB	Plutonium 238	HYDRIC	0.001	pCi/g	J
MA01R	VE	BI00329EB	Plutonium 239/240	HYDRIC	0	pCi/g	U
MA01R	VE	BI00329EB	Radium 226	HYDRIC		pCi/g	
MA01R	VE	BI00329EB	Total Uranium	HYDRIC		pCi/g	
MA01R	VE	BI00331EB	Americium 241	HYDRIC	0.002	pCi/g	BJ
MA01R	VE	BI00331EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA01R	VE	BI00331EB	Plutonium 239/240	HYDRIC	0	pCi/g	U
MA01R	VE	BI00331EB	Radium 226	HYDRIC		pCi/g	
MA01R	VE	BI00331EB	Total Uranium	HYDRIC	0.047	pCi/g	JX
MA02A	SM	BI00173EB	Americium 241	HYDRIC	0	pCi/g	UX
MA02A	SM	BI00173EB	Plutonium 238	HYDRIC	0.001	pCi/g	U
MA02A	SM	BI00173EB	Plutonium 239/240	HYDRIC	0.003	pCi/g	U
MA02A	SM	BI00173EB	Radium 226	HYDRIC		pCi/g	
MA02A	SM	BI00173EB	Total Uranium	HYDRIC		pCi/g	
MA02A	VE	BI00311EB	Americium 241	HYDRIC	0.002	pCi/g	J
MA02A	VE	BI00311EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA02A	VE	BI00311EB	Plutonium 239/240	HYDRIC	0.008	pCi/g	J
MA02A	VE	BI00311EB	Radium 226	HYDRIC		pCi/g	
MA02A	VE	BI00311EB	Total Uranium	HYDRIC	1.3	pCi/g	
MA02R	SM	BI00248EB	Americium 241	HYDRIC	0	pCi/g	U
MA02R	SM	BI00248EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA02R	SM	BI00248EB	Plutonium 239/240	HYDRIC	0.003	pCi/g	J
MA02R	SM	BI00248EB	Radium 226	HYDRIC		pCi/g	
MA02R	SM	BI00248EB	Total Uranium	HYDRIC	0.091	pCi/g	J

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MA03A	SM	BI00214EB	Americium 241	HYDRIC	0	pCi/g	U
MA03A	SM	BI00214EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA03A	SM	BI00214EB	Plutonium 239/240	HYDRIC	0.008	pCi/g	J
MA03A	SM	BI00214EB	Radium 226	HYDRIC		pCi/g	
MA03A	SM	BI00214EB	Total Uranium	HYDRIC		pCi/g	
MA03A	VE	BI00323EB	Americium 241	HYDRIC	-0.001	pCi/g	U
MA03A	VE	BI00323EB	Americium 241	HYDRIC	0.001	pCi/g	U
MA03A	VE	BI00323EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA03A	VE	BI00323EB	Plutonium 238	HYDRIC	0.001	pCi/g	U
MA03A	VE	BI00323EB	Plutonium 239/240	HYDRIC	0.004	pCi/g	J
MA03A	VE	BI00323EB	Plutonium 239/240	HYDRIC	0.003	pCi/g	J
MA03A	VE	BI00323EB	Radium 226	HYDRIC		pCi/g	
MA03A	VE	BI00323EB	Radium 226	HYDRIC		pCi/g	
MA03A	VE	BI00323EB	Total Uranium	HYDRIC		pCi/g	
MA03A	VE	BI00323EB	Total Uranium	HYDRIC		pCi/g	
MA03R	SM	BI00246EB	Americium 241	HYDRIC	0.001	pCi/g	J
MA03R	SM	BI00246EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA03R	SM	BI00246EB	Plutonium 239/240	HYDRIC	0.001	pCi/g	J
MA03R	SM	BI00246EB	Radium 226	HYDRIC		pCi/g	
MA03R	SM	BI00246EB	Total Uranium	HYDRIC	0.088	pCi/g	J
MA04A	SM	BI00239EB	Americium 241	HYDRIC	0.002	pCi/g	J
MA04A	SM	BI00239EB	Plutonium 238	HYDRIC	-0.001	pCi/g	U
MA04A	SM	BI00239EB	Plutonium 239/240	HYDRIC	0.008	pCi/g	J
MA04A	SM	BI00239EB	Radium 226	HYDRIC		pCi/g	
MA04A	SM	BI00239EB	Total Uranium	HYDRIC	0.12	pCi/g	J
MA04A	TA	BI00241EB	Americium 241	HYDRIC	0.005	pCi/g	U
MA04A	TA	BI00241EB	Plutonium 238	HYDRIC	0.006	pCi/g	U
MA04A	TA	BI00241EB	Plutonium 239/240	HYDRIC	0.012	pCi/g	J
MA04A	TA	BI00241EB	Radium 226	HYDRIC		pCi/g	
MA04A	TA	BI00241EB	Total Uranium	HYDRIC		pCi/g	
MA04A	VE	BI00325EB	Americium 241	HYDRIC	0.001	pCi/g	U
MA04A	VE	BI00325EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA04A	VE	BI00325EB	Plutonium 239/240	HYDRIC	0.001	pCi/g	J
MA04A	VE	BI00325EB	Radium 226	HYDRIC		pCi/g	
MA04A	VE	BI00325EB	Total Uranium	HYDRIC		pCi/g	
MA04R	SM	BI00240EB	Americium 241	HYDRIC	0.001	pCi/g	J
MA04R	SM	BI00240EB	Plutonium 238	HYDRIC	0	pCi/g	U
MA04R	SM	BI00240EB	Plutonium 239/240	HYDRIC	0.001	pCi/g	J
MA04R	SM	BI00240EB	Radium 226	HYDRIC		pCi/g	
MA04R	SM	BI00240EB	Total Uranium	HYDRIC	0.082	pCi/g	J
MD01A	SM	BI00190EB	Americium 241	MESIC	0	pCi/g	U
MD01A	SM	BI00190EB	Plutonium 238	MESIC	0.006	pCi/g	U
MD01A	SM	BI00190EB	Plutonium 239/240	MESIC	0.026	pCi/g	
MD01A	SM	BI00190EB	Radium 226	MESIC		pCi/g	
MD01A	SM	BI00190EB	Total Uranium	MESIC		pCi/g	
MD01A	TA	BI00262EB	Americium 241	MESIC	0.002	pCi/g	U
MD01A	TA	BI00262EB	Plutonium 238	MESIC	0.008	pCi/g	U
MD01A	TA	BI00262EB	Plutonium 239/240	MESIC	0.027	pCi/g	
MD01A	TA	BI00262EB	Radium 226	MESIC		pCi/g	
MD01A	TA	BI00262EB	Total Uranium	MESIC		pCi/g	
MD01A	VE	BI00308EB	Americium 241	MESIC	0.001	pCi/g	BJ
MD01A	VE	BI00308EB	Plutonium 238	MESIC	0	pCi/g	U
MD01A	VE	BI00308EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MD01A	VE	BI00308EB	Radium 226	MESIC		pCi/g	
MD01A	VE	BI00308EB	Total Uranium	MESIC	0.057	pCi/g	JX

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MD01A	VE	BI00310EB	Americium 241	MESIC	-0.001	pCi/g	U
MD01A	VE	BI00310EB	Plutonium 238	MESIC	0	pCi/g	U
MD01A	VE	BI00310EB	Plutonium 239/240	MESIC	0	pCi/g	U
MD01A	VE	BI00310EB	Radium 226	MESIC		pCi/g	
MD01A	VE	BI00310EB	Total Uranium	MESIC	0.043	pCi/g	JX
MD01B	SM	BI00269EB	Americium 241	MESIC	0.001	pCi/g	U
MD01B	SM	BI00269EB	Plutonium 238	MESIC	0	pCi/g	U
MD01B	SM	BI00269EB	Plutonium 239/240	MESIC	0	pCi/g	U
MD01B	SM	BI00269EB	Radium 226	MESIC		pCi/g	
MD01B	SM	BI00269EB	Total Uranium	MESIC	0.084	pCi/g	J
MD02A	SM	BI00187EB	Americium 241	MESIC	0.001	pCi/g	U
MD02A	SM	BI00187EB	Plutonium 238	MESIC	0	pCi/g	U
MD02A	SM	BI00187EB	Plutonium 239/240	MESIC	0.003	pCi/g	J
MD02A	SM	BI00187EB	Radium 226	MESIC		pCi/g	
MD02A	SM	BI00187EB	Total Uranium	MESIC		pCi/g	
MD02A	VE	BI00287EB	Americium 241	MESIC	0.001	pCi/g	U
MD02A	VE	BI00287EB	Plutonium 238	MESIC	0	pCi/g	U
MD02A	VE	BI00287EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MD02A	VE	BI00287EB	Radium 226	MESIC		pCi/g	
MD02A	VE	BI00287EB	Total Uranium	MESIC		pCi/g	
MD02A	VE	BI00288EB	Americium 241	MESIC	0.002	pCi/g	U
MD02A	VE	BI00288EB	Plutonium 238	MESIC	0.002	pCi/g	J
MD02A	VE	BI00288EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MD02A	VE	BI00288EB	Radium 226	MESIC		pCi/g	
MD02A	VE	BI00288EB	Total Uranium	MESIC	0.039	pCi/g	JX
MG01A	SM	BI00366EB	Americium 241	MESIC	0	pCi/g	U
MG01A	SM	BI00366EB	Plutonium 238	MESIC	0	pCi/g	U
MG01A	SM	BI00366EB	Plutonium 239/240	MESIC	0.003	pCi/g	J
MG01A	SM	BI00366EB	Radium 226	MESIC		pCi/g	
MG01A	SM	BI00366EB	Total Uranium	MESIC	0.090	pCi/g	J
MG01R	SM	BI00267EB	Americium 241	MESIC	-0.002	pCi/g	U
MG01R	SM	BI00267EB	Plutonium 238	MESIC	0.001	pCi/g	U
MG01R	SM	BI00267EB	Plutonium 239/240	MESIC	0	pCi/g	U
MG01R	SM	BI00267EB	Radium 226	MESIC		pCi/g	
MG01R	SM	BI00267EB	Total Uranium	MESIC	0.12	pCi/g	J
MG02A	VE	BI00318EB	Americium 241	MESIC	0.002	pCi/g	BJ
MG02A	VE	BI00318EB	Plutonium 238	MESIC	0	pCi/g	U
MG02A	VE	BI00318EB	Plutonium 239/240	MESIC	0.001	pCi/g	U
MG02A	VE	BI00318EB	Radium 226	MESIC		pCi/g	
MG02A	VE	BI00318EB	Total Uranium	MESIC	0.054	pCi/g	JX
MG02A	VE	BI00319EB	Americium 241	MESIC	0.001	pCi/g	BJ
MG02A	VE	BI00319EB	Plutonium 238	MESIC	0	pCi/g	U
MG02A	VE	BI00319EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG02A	VE	BI00319EB	Radium 226	MESIC		pCi/g	
MG02A	VE	BI00319EB	Total Uranium	MESIC	0.036	pCi/g	JX
MG02A	VE	BI00520EB	Americium 241	MESIC	0.002	pCi/g	BJ
MG02A	VE	BI00520EB	Plutonium 238	MESIC	0	pCi/g	U
MG02A	VE	BI00520EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG02A	VE	BI00520EB	Radium 226	MESIC		pCi/g	
MG02A	VE	BI00520EB	Total Uranium	MESIC		pCi/g	
MG02A	VE	BI00521EB	Americium 241	MESIC	0.001	pCi/g	BJ
MG02A	VE	BI00521EB	Plutonium 238	MESIC	0	pCi/g	U
MG02A	VE	BI00521EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG02A	VE	BI00521EB	Radium 226	MESIC		pCi/g	
MG02A	VE	BI00521EB	Total Uranium	MESIC	0.042	pCi/g	JX

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MG02R	SM	BI00266EB	Americium 241	MESIC	0	pCi/g	U
MG02R	SM	BI00266EB	Plutonium 238	MESIC	-0.001	pCi/g	U
MG02R	SM	BI00266EB	Plutonium 239/240	MESIC	0	pCi/g	U
MG02R	SM	BI00266EB	Radium 226	MESIC		pCi/g	
MG02R	SM	BI00266EB	Total Uranium	MESIC	0.11	pCi/g	J
MG03A	SM	BI00365EB	Americium 241	MESIC	0.003	pCi/g	U
MG03A	SM	BI00365EB	Plutonium 238	MESIC	0.004	pCi/g	U
MG03A	SM	BI00365EB	Plutonium 239/240	MESIC	0.010	pCi/g	U
MG03A	SM	BI00365EB	Radium 226	MESIC		pCi/g	
MG03A	SM	BI00365EB	Total Uranium	MESIC		pCi/g	
MG03A	TA	BI00249EB	Americium 241	MESIC	0.021	pCi/g	
MG03A	TA	BI00249EB	Plutonium 238	MESIC	-0.002	pCi/g	U
MG03A	TA	BI00249EB	Plutonium 239/240	MESIC	0.033	pCi/g	
MG03A	TA	BI00249EB	Radium 226	MESIC		pCi/g	
MG03A	TA	BI00249EB	Total Uranium	MESIC		pCi/g	
MG03A	VE	BI00341EB	Americium 241	MESIC	0.013	pCi/g	BJ
MG03A	VE	BI00341EB	Plutonium 238	MESIC	0.002	pCi/g	J
MG03A	VE	BI00341EB	Plutonium 239/240	MESIC	0.048	pCi/g	B
MG03A	VE	BI00341EB	Radium 226	MESIC		pCi/g	
MG03A	VE	BI00341EB	Total Uranium	MESIC		pCi/g	
MG03A	VE	BI00517EB	Americium 241	MESIC	0.011	pCi/g	BJ
MG03A	VE	BI00517EB	Plutonium 238	MESIC	0	pCi/g	U
MG03A	VE	BI00517EB	Plutonium 239/240	MESIC	0.054	pCi/g	B
MG03A	VE	BI00517EB	Radium 226	MESIC		pCi/g	
MG03A	VE	BI00517EB	Total Uranium	MESIC	0.046	pCi/g	JX
MG03A	VE	BI00518EB	Americium 241	MESIC	0.006	pCi/g	BJ
MG03A	VE	BI00518EB	Plutonium 238	MESIC	0	pCi/g	U
MG03A	VE	BI00518EB	Plutonium 239/240	MESIC	0.022	pCi/g	B
MG03A	VE	BI00518EB	Radium 226	MESIC		pCi/g	
MG03A	VE	BI00518EB	Total Uranium	MESIC		pCi/g	
MG03A	VE	BI00519EB	Americium 241	MESIC	0.014	pCi/g	BJ
MG03A	VE	BI00519EB	Plutonium 238	MESIC	0.002	pCi/g	J
MG03A	VE	BI00519EB	Plutonium 239/240	MESIC	0.045	pCi/g	B
MG03A	VE	BI00519EB	Radium 226	MESIC		pCi/g	
MG03A	VE	BI00519EB	Total Uranium	MESIC	0.040	pCi/g	JX
MG03R	TA	BI00301EB	Americium 241	MESIC	0	pCi/g	U
MG03R	TA	BI00301EB	Plutonium 238	MESIC	0	pCi/g	U
MG03R	TA	BI00301EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG03R	TA	BI00301EB	Radium 226	MESIC		pCi/g	
MG03R	TA	BI00301EB	Total Uranium	MESIC		pCi/g	
MG03R	VE	BI00333EB	Americium 241	MESIC	0.001	pCi/g	BJ
MG03R	VE	BI00333EB	Plutonium 238	MESIC	0	pCi/g	U
MG03R	VE	BI00333EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG03R	VE	BI00333EB	Radium 226	MESIC		pCi/g	
MG03R	VE	BI00333EB	Total Uranium	MESIC	0.056	pCi/g	JX
MG03R	VE	BI00336EB	Americium 241	MESIC	0	pCi/g	U
MG03R	VE	BI00336EB	Plutonium 238	MESIC	0	pCi/g	U
MG03R	VE	BI00336EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG03R	VE	BI00336EB	Radium 226	MESIC		pCi/g	
MG03R	VE	BI00336EB	Total Uranium	MESIC	0.036	pCi/g	JX
MG04A	SM	BI00353EB	Americium 241	MESIC	0.086	pCi/g	
MG04A	SM	BI00353EB	Plutonium 238	MESIC	0.008	pCi/g	J
MG04A	SM	BI00353EB	Plutonium 239/240	MESIC	0.47	pCi/g	
MG04A	SM	BI00353EB	Radium 226	MESIC		pCi/g	
MG04A	SM	BI00353EB	Total Uranium	MESIC	0.11	pCi/g	J

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MG04A	VE	BI00255EB	Americium 241	MESIC	0.023	pCi/g	B
MG04A	VE	BI00255EB	Plutonium 238	MESIC	0.002	pCi/g	J
MG04A	VE	BI00255EB	Plutonium 239/240	MESIC	0.091	pCi/g	B
MG04A	VE	BI00255EB	Radium 226	MESIC		pCi/g	
MG04A	VE	BI00255EB	Total Uranium	MESIC	0.049	pCi/g	JX
MG04A	VE	BI00256EB	Americium 241	MESIC	0.011	pCi/g	BJ
MG04A	VE	BI00256EB	Plutonium 238	MESIC	0	pCi/g	U
MG04A	VE	BI00256EB	Plutonium 239/240	MESIC	0.051	pCi/g	B
MG04A	VE	BI00256EB	Radium 226	MESIC		pCi/g	
MG04A	VE	BI00256EB	Total Uranium	MESIC	0.045	pCi/g	JX
MG04A	VE	BI00522EB	Americium 241	MESIC	0.12	pCi/g	B
MG04A	VE	BI00522EB	Americium 241	MESIC	0.088	pCi/g	B
MG04A	VE	BI00522EB	Plutonium 238	MESIC	0.001	pCi/g	J
MG04A	VE	BI00522EB	Plutonium 238	MESIC	0.001	pCi/g	J
MG04A	VE	BI00522EB	Plutonium 239/240	MESIC	0.11	pCi/g	B
MG04A	VE	BI00522EB	Plutonium 239/240	MESIC	0.12	pCi/g	B
MG04A	VE	BI00522EB	Radium 226	MESIC		pCi/g	
MG04A	VE	BI00522EB	Radium 226	MESIC	11	pCi/g	
MG04A	VE	BI00522EB	Total Uranium	MESIC	0.038	pCi/g	JX
MG04A	VE	BI00522EB	Total Uranium	MESIC	0.038	pCi/g	JX
MG04A	VE	BI00523EB	Americium 241	MESIC	0.003	pCi/g	BJ
MG04A	VE	BI00523EB	Plutonium 238	MESIC	0	pCi/g	U
MG04A	VE	BI00523EB	Plutonium 239/240	MESIC	0.014	pCi/g	BJ
MG04A	VE	BI00523EB	Radium 226	MESIC		pCi/g	
MG04A	VE	BI00523EB	Total Uranium	MESIC	0.040	pCi/g	JX
MG04R	SM	BI00271EB	Americium 241	MESIC	0	pCi/g	U
MG04R	SM	BI00271EB	Plutonium 238	MESIC	-0.001	pCi/g	U
MG04R	SM	BI00271EB	Plutonium 239/240	MESIC	0	pCi/g	U
MG04R	SM	BI00271EB	Radium 226	MESIC		pCi/g	
MG04R	SM	BI00271EB	Total Uranium	MESIC	0.14	pCi/g	J
MG04R	VE	BI00280EB	Americium 241	MESIC	-0.001	pCi/g	U
MG04R	VE	BI00280EB	Plutonium 238	MESIC	0	pCi/g	U
MG04R	VE	BI00280EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG04R	VE	BI00280EB	Radium 226	MESIC		pCi/g	
MG04R	VE	BI00280EB	Total Uranium	MESIC	0.054	pCi/g	JX
MG04R	VE	BI00281EB	Americium 241	MESIC	0.001	pCi/g	U
MG04R	VE	BI00281EB	Americium 241	MESIC	-0.001	pCi/g	U
MG04R	VE	BI00281EB	Plutonium 238	MESIC	0	pCi/g	U
MG04R	VE	BI00281EB	Plutonium 238	MESIC	0	pCi/g	U
MG04R	VE	BI00281EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG04R	VE	BI00281EB	Plutonium 239/240	MESIC	0.001	pCi/g	BJ
MG04R	VE	BI00281EB	Radium 226	MESIC		pCi/g	
MG04R	VE	BI00281EB	Radium 226	MESIC		pCi/g	
MG04R	VE	BI00281EB	Total Uranium	MESIC	0.046	pCi/g	JX
MG04R	VE	BI00281EB	Total Uranium	MESIC	0.037	pCi/g	JX
MR01A	VE	BI00363EB	Americium 241	MESIC	0	pCi/g	U
MR01A	VE	BI00363EB	Plutonium 238	MESIC	0	pCi/g	U
MR01A	VE	BI00363EB	Plutonium 239/240	MESIC	0	pCi/g	U
MR01A	VE	BI00363EB	Radium 226	MESIC		pCi/g	
MR01A	VE	BI00363EB	Total Uranium	MESIC		pCi/g	
MR02A	SM	BI00364EB	Americium 241	MESIC	0	pCi/g	U
MR02A	SM	BI00364EB	Plutonium 238	MESIC	0	pCi/g	U
MR02A	SM	BI00364EB	Plutonium 239/240	MESIC	0.004	pCi/g	J
MR02A	SM	BI00364EB	Radium 226	MESIC		pCi/g	
MR02A	SM	BI00364EB	Total Uranium	MESIC	0.26	pCi/g	



## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MR02A	VE	BI00320EB	Americium 241	MESIC	0.003	pCi/g	BJ
MR02A	VE	BI00320EB	Plutonium 238	MESIC	0	pCi/g	U
MR02A	VE	BI00320EB	Plutonium 239/240	MESIC	0.006	pCi/g	BJ
MR02A	VE	BI00320EB	Radium 226	MESIC		pCi/g	
MR02A	VE	BI00320EB	Total Uranium	MESIC	0.056	pCi/g	JX
MR03A	TA	BI00302EB	Americium 241	MESIC	0.002	pCi/g	U
MR03A	TA	BI00302EB	Plutonium 238	MESIC	0.001	pCi/g	U
MR03A	TA	BI00302EB	Plutonium 239/240	MESIC	0.001	pCi/g	U
MR03A	TA	BI00302EB	Radium 226	MESIC		pCi/g	
MR03A	TA	BI00302EB	Total Uranium	MESIC		pCi/g	
MR03A	VE	BI00282EB	Americium 241	MESIC	0.005	pCi/g	BJ
MR03A	VE	BI00282EB	Plutonium 238	MESIC	0.001	pCi/g	J
MR03A	VE	BI00282EB	Plutonium 239/240	MESIC	0.001	pCi/g	U
MR03A	VE	BI00282EB	Radium 226	MESIC		pCi/g	
MR03A	VE	BI00282EB	Total Uranium	MESIC	0.086	pCi/g	JX
MR03A	VE	BI00283EB	Americium 241	MESIC	0.001	pCi/g	U
MR03A	VE	BI00283EB	Plutonium 238	MESIC	0	pCi/g	U
MR03A	VE	BI00283EB	Plutonium 239/240	MESIC	0	pCi/g	U
MR03A	VE	BI00283EB	Radium 226	MESIC		pCi/g	
MR03A	VE	BI00283EB	Total Uranium	MESIC		pCi/g	
MR04A	SM	BI00354EB	Americium 241	MESIC	0.001	pCi/g	U
MR04A	SM	BI00354EB	Plutonium 238	MESIC	0	pCi/g	U
MR04A	SM	BI00354EB	Plutonium 239/240	MESIC	0.003	pCi/g	J
MR04A	SM	BI00354EB	Radium 226	MESIC		pCi/g	
MR04A	SM	BI00354EB	Total Uranium	MESIC	0.099	pCi/g	J
MR04A	VE	BI00259EB	Americium 241	MESIC	0.003	pCi/g	BJ
MR04A	VE	BI00259EB	Plutonium 238	MESIC	0.001	pCi/g	J
MR04A	VE	BI00259EB	Plutonium 239/240	MESIC	0.007	pCi/g	BJ
MR04A	VE	BI00259EB	Radium 226	MESIC		pCi/g	
MR04A	VE	BI00259EB	Total Uranium	MESIC	0.040	pCi/g	JX
MR04A	VE	BI00261EB	Americium 241	MESIC	0.001	pCi/g	U
MR04A	VE	BI00261EB	Plutonium 238	MESIC	0	pCi/g	U
MR04A	VE	BI00261EB	Plutonium 239/240	MESIC	0	pCi/g	U
MR04A	VE	BI00261EB	Radium 226	MESIC		pCi/g	
MR04A	VE	BI00261EB	Total Uranium	MESIC		pCi/g	
MR04A	VE	BI00515EB	Americium 241	MESIC	-0.001	pCi/g	U
MR04A	VE	BI00515EB	Plutonium 238	MESIC	0	pCi/g	U
MR04A	VE	BI00515EB	Plutonium 239/240	MESIC	0.001	pCi/g	U
MR04A	VE	BI00515EB	Radium 226	MESIC		pCi/g	
MR04A	VE	BI00515EB	Total Uranium	MESIC		pCi/g	
MR04A	VE	BI00516EB	Americium 241	MESIC	0.022	pCi/g	BX
MR04A	VE	BI00516EB	Americium 241	MESIC	0.005	pCi/g	BJ
MR04A	VE	BI00516EB	Plutonium 238	MESIC	0	pCi/g	U
MR04A	VE	BI00516EB	Plutonium 238	MESIC	0	pCi/g	U
MR04A	VE	BI00516EB	Plutonium 239/240	MESIC	0.006	pCi/g	BJ
MR04A	VE	BI00516EB	Plutonium 239/240	MESIC	0.008	pCi/g	BJ
MR04A	VE	BI00516EB	Radium 226	MESIC	11	pCi/g	
MR04A	VE	BI00516EB	Radium 226	MESIC		pCi/g	
MR04A	VE	BI00516EB	Total Uranium	MESIC	0.064	pCi/g	JX
MR04A	VE	BI00516EB	Total Uranium	MESIC	0.053	pCi/g	JX
MW01A	SM	BI00189EB	Americium 241	HYDRIC	0.002	pCi/g	J
MW01A	SM	BI00189EB	Plutonium 238	HYDRIC	0.001	pCi/g	U
MW01A	SM	BI00189EB	Plutonium 239/240	HYDRIC	0	pCi/g	U
MW01A	SM	BI00189EB	Radium 226	HYDRIC		pCi/g	
MW01A	SM	BI00189EB	Total Uranium	HYDRIC		pCi/g	

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MW01A	TA	BI00300EB	Americium 241	HYDRIC	-0.001	pCi/g	U
MW01A	TA	BI00300EB	Plutonium 238	HYDRIC	0.004	pCi/g	U
MW01A	TA	BI00300EB	Plutonium 239/240	HYDRIC	0.003	pCi/g	U
MW01A	TA	BI00300EB	Radium 226	HYDRIC		pCi/g	
MW01A	TA	BI00300EB	Total Uranium	HYDRIC	0.042	pCi/g	J
MW01A	VE	BI00293EB	Americium 241	HYDRIC	-0.001	pCi/g	U
MW01A	VE	BI00293EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW01A	VE	BI00293EB	Plutonium 239/240	HYDRIC	0.002	pCi/g	J
MW01A	VE	BI00293EB	Radium 226	HYDRIC		pCi/g	
MW01A	VE	BI00293EB	Total Uranium	HYDRIC	0.059	pCi/g	J
MW01R	SM	BI00242EB	Americium 241	HYDRIC	0	pCi/g	U
MW01R	SM	BI00242EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW01R	SM	BI00242EB	Plutonium 239/240	HYDRIC	0.002	pCi/g	J
MW01R	SM	BI00242EB	Radium 226	HYDRIC		pCi/g	
MW01R	SM	BI00242EB	Total Uranium	HYDRIC	0.090	pCi/g	J
MW02A	SM	BI00232EB	Americium 241	HYDRIC	0.002	pCi/g	J
MW02A	SM	BI00232EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW02A	SM	BI00232EB	Plutonium 239/240	HYDRIC	0	pCi/g	U
MW02A	SM	BI00232EB	Radium 226	HYDRIC		pCi/g	
MW02A	SM	BI00232EB	Total Uranium	HYDRIC	0.096	pCi/g	J
MW02R	SM	BI00243EB	Americium 241	HYDRIC	0.001	pCi/g	J
MW02R	SM	BI00243EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW02R	SM	BI00243EB	Plutonium 239/240	HYDRIC	0.001	pCi/g	J
MW02R	SM	BI00243EB	Radium 226	HYDRIC		pCi/g	
MW02R	SM	BI00243EB	Total Uranium	HYDRIC	0.10	pCi/g	J
MW03A	HE	BI00174EB	Americium 241	HYDRIC	0.009	pCi/g	J
MW03A	HE	BI00174EB	Plutonium 238	HYDRIC	0.001	pCi/g	U
MW03A	HE	BI00174EB	Plutonium 239/240	HYDRIC	0.051	pCi/g	B
MW03A	HE	BI00174EB	Radium 226	HYDRIC		pCi/g	
MW03A	HE	BI00174EB	Total Uranium	HYDRIC	0.28	pCi/g	
MW03A	SM	BI00188EB	Americium 241	HYDRIC	0.001	pCi/g	U
MW03A	SM	BI00188EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW03A	SM	BI00188EB	Plutonium 239/240	HYDRIC	0.001	pCi/g	U
MW03A	SM	BI00188EB	Radium 226	HYDRIC		pCi/g	
MW03A	SM	BI00188EB	Total Uranium	HYDRIC		pCi/g	
MW03A	VE	BI00312EB	Americium 241	HYDRIC	0	pCi/g	U
MW03A	VE	BI00312EB	Plutonium 238	HYDRIC	-0.001	pCi/g	U
MW03A	VE	BI00312EB	Plutonium 239/240	HYDRIC	0.002	pCi/g	J
MW03A	VE	BI00312EB	Radium 226	HYDRIC		pCi/g	
MW03A	VE	BI00312EB	Total Uranium	HYDRIC		pCi/g	
MW03A	VE	BI00313EB	Americium 241	HYDRIC	0.002	pCi/g	J
MW03A	VE	BI00313EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW03A	VE	BI00313EB	Plutonium 239/240	HYDRIC	0.011	pCi/g	J
MW03A	VE	BI00313EB	Radium 226	HYDRIC		pCi/g	
MW03A	VE	BI00313EB	Total Uranium	HYDRIC		pCi/g	
MW03R	SM	BI00244EB	Americium 241	HYDRIC	-0.001	pCi/g	U
MW03R	SM	BI00244EB	Plutonium 238	HYDRIC	0.001	pCi/g	J
MW03R	SM	BI00244EB	Plutonium 239/240	HYDRIC	0.003	pCi/g	J
MW03R	SM	BI00244EB	Radium 226	HYDRIC		pCi/g	
MW03R	SM	BI00244EB	Total Uranium	HYDRIC	0.096	pCi/g	J
MW03R	TA	BI00327EB	Americium 241	HYDRIC	0.002	pCi/g	U
MW03R	TA	BI00327EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW03R	TA	BI00327EB	Plutonium 239/240	HYDRIC	0.002	pCi/g	U
MW03R	TA	BI00327EB	Radium 226	HYDRIC		pCi/g	
MW03R	TA	BI00327EB	Total Uranium	HYDRIC		pCi/g	

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
MW03R	VE	BI00304EB	Americium 241	HYDRIC	0.001	pCi/g	U
MW03R	VE	BI00304EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW03R	VE	BI00304EB	Plutonium 239/240	HYDRIC	0	pCi/g	U
MW03R	VE	BI00304EB	Radium 226	HYDRIC		pCi/g	
MW03R	VE	BI00304EB	Total Uranium	HYDRIC		pCi/g	
MW03R	VE	BI00305EB	Americium 241	HYDRIC	0.002	pCi/g	J
MW03R	VE	BI00305EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW03R	VE	BI00305EB	Plutonium 239/240	HYDRIC	0.001	pCi/g	J
MW03R	VE	BI00305EB	Radium 226	HYDRIC		pCi/g	
MW03R	VE	BI00305EB	Total Uranium	HYDRIC		pCi/g	
MW04A	SM	BI00186EB	Americium 241	HYDRIC	0.001	pCi/g	U
MW04A	SM	BI00186EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW04A	SM	BI00186EB	Plutonium 239/240	HYDRIC	0.001	pCi/g	U
MW04A	SM	BI00186EB	Radium 226	HYDRIC		pCi/g	
MW04A	SM	BI00186EB	Total Uranium	HYDRIC		pCi/g	
MW04A	VE	BI00344EB	Americium 241	HYDRIC	0.001	pCi/g	J
MW04A	VE	BI00344EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW04A	VE	BI00344EB	Plutonium 239/240	HYDRIC	0.003	pCi/g	J
MW04A	VE	BI00344EB	Radium 226	HYDRIC		pCi/g	
MW04A	VE	BI00344EB	Total Uranium	HYDRIC		pCi/g	
MW04A	VE	BI00345EB	Americium 241	HYDRIC	0.001	pCi/g	U
MW04A	VE	BI00345EB	Plutonium 238	HYDRIC	0	pCi/g	U
MW04A	VE	BI00345EB	Plutonium 239/240	HYDRIC	0.002	pCi/g	J
MW04A	VE	BI00345EB	Radium 226	HYDRIC		pCi/g	
MW04A	VE	BI00345EB	Total Uranium	HYDRIC		pCi/g	
MX01R	HE	BI00299EB	Americium 241	XERIC	0	pCi/g	U
MX01R	HE	BI00299EB	Americium 241	XERIC	0	pCi/g	U
MX01R	HE	BI00299EB	Plutonium 238	XERIC	0	pCi/g	U
MX01R	HE	BI00299EB	Plutonium 238	XERIC	0.002	pCi/g	U
MX01R	HE	BI00299EB	Plutonium 239/240	XERIC	0.003	pCi/g	BJ
MX01R	HE	BI00299EB	Plutonium 239/240	XERIC	0.007	pCi/g	BJ
MX01R	HE	BI00299EB	Radium 226	XERIC		pCi/g	
MX01R	HE	BI00299EB	Radium 226	XERIC		pCi/g	
MX01R	HE	BI00299EB	Total Uranium	XERIC	0.14	pCi/g	J
MX01R	HE	BI00299EB	Total Uranium	XERIC	0.13	pCi/g	J
MX01R	SM	BI00268EB	Americium 241	XERIC	0.001	pCi/g	U
MX01R	SM	BI00268EB	Plutonium 238	XERIC	0.003	pCi/g	U
MX01R	SM	BI00268EB	Plutonium 239/240	XERIC	0.003	pCi/g	J
MX01R	SM	BI00268EB	Radium 226	XERIC		pCi/g	
MX01R	SM	BI00268EB	Total Uranium	XERIC	0.068	pCi/g	J
MX02R	SM	BI00263EB	Americium 241	XERIC	-0.001	pCi/g	U
MX02R	SM	BI00263EB	Plutonium 238	XERIC	0	pCi/g	U
MX02R	SM	BI00263EB	Plutonium 239/240	XERIC	0	pCi/g	U
MX02R	SM	BI00263EB	Radium 226	XERIC		pCi/g	
MX02R	SM	BI00263EB	Total Uranium	XERIC	0.11	pCi/g	J
MX03R	SM	BI00265EB	Americium 241	XERIC	0.001	pCi/g	U
MX03R	SM	BI00265EB	Plutonium 238	XERIC	0	pCi/g	U
MX03R	SM	BI00265EB	Plutonium 239/240	XERIC	0	pCi/g	U
MX03R	SM	BI00265EB	Radium 226	XERIC		pCi/g	
MX03R	SM	BI00265EB	Total Uranium	XERIC	0.13	pCi/g	J
RCSP01	FI	BI00504EB	Americium 241	AQUATIC	0	pCi/g	U
RCSP01	FI	BI00504EB	Plutonium 238	AQUATIC	0	pCi/g	U
RCSP01	FI	BI00504EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	J
RCSP01	FI	BI00504EB	Radium 226	AQUATIC		pCi/g	
RCSP01	FI	BI00504EB	Total Uranium	AQUATIC	0.57	pCi/g	

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
RCSP01	FI	BI00505EB	Americium 241	AQUATIC	0.001	pCi/g	U
RCSP01	FI	BI00505EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
RCSP01	FI	BI00505EB	Plutonium 239/240	AQUATIC	0.007	pCi/g	J
RCSP01	FI	BI00505EB	Radium 226	AQUATIC		pCi/g	
RCSP01	FI	BI00505EB	Total Uranium	AQUATIC	0.62	pCi/g	
RCSP01	FI	BI00506EB	Americium 241	AQUATIC	0.001	pCi/g	U
RCSP01	FI	BI00506EB	Americium 241	AQUATIC	0.003	pCi/g	U
RCSP01	FI	BI00506EB	Plutonium 238	AQUATIC	0	pCi/g	U
RCSP01	FI	BI00506EB	Plutonium 238	AQUATIC	0	pCi/g	U
RCSP01	FI	BI00506EB	Plutonium 239/240	AQUATIC	0.020	pCi/g	J
RCSP01	FI	BI00506EB	Plutonium 239/240	AQUATIC	0.003	pCi/g	U
RCSP01	FI	BI00506EB	Radium 226	AQUATIC		pCi/g	
RCSP01	FI	BI00506EB	Radium 226	AQUATIC		pCi/g	
RCSP01	FI	BI00506EB	Total Uranium	AQUATIC	0.38	pCi/g	
RCSP01	FI	BI00506EB	Total Uranium	AQUATIC	0.40	pCi/g	
SW003	HE	BI00185EB	Americium 241	AQUATIC	0	pCi/g	U
SW003	HE	BI00185EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW003	HE	BI00185EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SW003	HE	BI00185EB	Radium 226	AQUATIC		pCi/g	
SW003	HE	BI00185EB	Total Uranium	AQUATIC	0.12	pCi/g	J
SW005	FI	BI00275EB	Americium 241	AQUATIC	0	pCi/g	U
SW005	FI	BI00275EB	Americium 241	AQUATIC	0	pCi/g	U
SW005	FI	BI00275EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW005	FI	BI00275EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW005	FI	BI00275EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SW005	FI	BI00275EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	J
SW005	FI	BI00275EB	Radium 226	AQUATIC		pCi/g	
SW005	FI	BI00275EB	Radium 226	AQUATIC		pCi/g	
SW005	FI	BI00275EB	Total Uranium	AQUATIC		pCi/g	X
SW005	FI	BI00275EB	Total Uranium	AQUATIC		pCi/g	X
SW005	FI	BI00276EB	Americium 241	AQUATIC	0	pCi/g	U
SW005	FI	BI00276EB	Americium 241	AQUATIC	0.001	pCi/g	J
SW005	FI	BI00276EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW005	FI	BI00276EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW005	FI	BI00276EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SW005	FI	BI00276EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
SW005	FI	BI00276EB	Radium 226	AQUATIC		pCi/g	
SW005	FI	BI00276EB	Radium 226	AQUATIC		pCi/g	
SW005	FI	BI00276EB	Total Uranium	AQUATIC		pCi/g	X
SW005	FI	BI00276EB	Total Uranium	AQUATIC		pCi/g	X
SW005	FI	BI00277EB	Americium 241	AQUATIC	0	pCi/g	U
SW005	FI	BI00277EB	Americium 241	AQUATIC	-0.001	pCi/g	U
SW005	FI	BI00277EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW005	FI	BI00277EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW005	FI	BI00277EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SW005	FI	BI00277EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SW005	FI	BI00277EB	Radium 226	AQUATIC		pCi/g	
SW005	FI	BI00277EB	Radium 226	AQUATIC		pCi/g	
SW005	FI	BI00277EB	Total Uranium	AQUATIC	0.071	pCi/g	J
SW005	FI	BI00277EB	Total Uranium	AQUATIC	0.084	pCi/g	J
SW026	FI	BI00475EB	Americium 241	AQUATIC	0	pCi/g	U
SW026	FI	BI00475EB	Americium 241	AQUATIC	0.002	pCi/g	U
SW026	FI	BI00475EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW026	FI	BI00475EB	Plutonium 238	AQUATIC	0.003	pCi/g	J
SW026	FI	BI00475EB	Plutonium 239/240	AQUATIC	0.008	pCi/g	J

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SW026	FI	BI00475EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SW026	FI	BI00475EB	Radium 226	AQUATIC	8.5	pCi/g	J
SW026	FI	BI00475EB	Radium 226	AQUATIC		pCi/g	
SW026	FI	BI00475EB	Total Uranium	AQUATIC	0.10	pCi/g	J
SW026	FI	BI00475EB	Total Uranium	AQUATIC	0.071	pCi/g	J
SW026	FI	BI00476EB	Americium 241	AQUATIC	0.003	pCi/g	J
SW026	FI	BI00476EB	Plutonium 238	AQUATIC	0.010	pCi/g	U
SW026	FI	BI00476EB	Plutonium 239/240	AQUATIC	0.015	pCi/g	J
SW026	FI	BI00476EB	Radium 226	AQUATIC		pCi/g	
SW026	FI	BI00476EB	Total Uranium	AQUATIC	0.14	pCi/g	J
SW026	FI	BI00477EB	Americium 241	AQUATIC	0	pCi/g	U
SW026	FI	BI00477EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
SW026	FI	BI00477EB	Plutonium 239/240	AQUATIC	0.003	pCi/g	J
SW026	FI	BI00477EB	Radium 226	AQUATIC		pCi/g	
SW026	FI	BI00477EB	Total Uranium	AQUATIC	0.24	pCi/g	
SW026	FI	BI00478EB	Americium 241	AQUATIC	0	pCi/g	U
SW026	FI	BI00478EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
SW026	FI	BI00478EB	Plutonium 239/240	AQUATIC	0.002	pCi/g	U
SW026	FI	BI00478EB	Radium 226	AQUATIC	24	pCi/g	
SW026	FI	BI00478EB	Total Uranium	AQUATIC	0.12	pCi/g	J
SW026	FI	BI00479EB	Americium 241	AQUATIC	0.011	pCi/g	U
SW026	FI	BI00479EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW026	FI	BI00479EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
SW026	FI	BI00479EB	Radium 226	AQUATIC		pCi/g	
SW026	FI	BI00479EB	Total Uranium	AQUATIC	0.28	pCi/g	
SW033	FI	BI00484EB	Americium 241	AQUATIC	0.004	pCi/g	U
SW033	FI	BI00484EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
SW033	FI	BI00484EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
SW033	FI	BI00484EB	Radium 226	AQUATIC		pCi/g	
SW033	FI	BI00484EB	Total Uranium	AQUATIC	0.25	pCi/g	
SW033	FI	BI00485EB	Americium 241	AQUATIC	0.002	pCi/g	U
SW033	FI	BI00485EB	Plutonium 238	AQUATIC	0	pCi/g	U
SW033	FI	BI00485EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
SW033	FI	BI00485EB	Radium 226	AQUATIC		pCi/g	
SW033	FI	BI00485EB	Total Uranium	AQUATIC	0.12	pCi/g	J
SW033	FI	BI00486EB	Americium 241	AQUATIC	0	pCi/g	U
SW033	FI	BI00486EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
SW033	FI	BI00486EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SW033	FI	BI00486EB	Radium 226	AQUATIC		pCi/g	
SW033	FI	BI00486EB	Total Uranium	AQUATIC	0.097	pCi/g	J
SW038	BM	BI00501EB	Americium 241	AQUATIC	-0.001	pCi/g	U
SW038	BM	BI00501EB	Plutonium 238	AQUATIC	-0.002	pCi/g	U
SW038	BM	BI00501EB	Plutonium 239/240	AQUATIC	0.018	pCi/g	J
SW038	BM	BI00501EB	Radium 226	AQUATIC		pCi/g	
SW038	BM	BI00501EB	Total Uranium	AQUATIC	0.12	pCi/g	J
SWC001	BM	BI00211EB	Americium 241	AQUATIC	0.002	pCi/g	U
SWC001	BM	BI00211EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
SWC001	BM	BI00211EB	Plutonium 239/240	AQUATIC	0.019	pCi/g	J
SWC001	BM	BI00211EB	Radium 226	AQUATIC		pCi/g	
SWC001	BM	BI00211EB	Total Uranium	AQUATIC		pCi/g	
SWC001	BM	BI00212EB	Americium 241	AQUATIC	0.004	pCi/g	U
SWC001	BM	BI00212EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	BM	BI00212EB	Plutonium 239/240	AQUATIC	0.032	pCi/g	
SWC001	BM	BI00212EB	Radium 226	AQUATIC		pCi/g	
SWC001	BM	BI00212EB	Total Uranium	AQUATIC		pCi/g	

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SWC001	BM	BI00500EB	Americium 241	AQUATIC	0.004	pCi/g	U
SWC001	BM	BI00500EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
SWC001	BM	BI00500EB	Plutonium 239/240	AQUATIC	0.006	pCi/g	U
SWC001	BM	BI00500EB	Radium 226	AQUATIC		pCi/g	
SWC001	BM	BI00500EB	Total Uranium	AQUATIC	0.10	pCi/g	J
SWC001	FI	BI00203EB	Americium 241	AQUATIC	0.015	pCi/g	U
SWC001	FI	BI00203EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
SWC001	FI	BI00203EB	Plutonium 239/240	AQUATIC	0.004	pCi/g	U
SWC001	FI	BI00203EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00203EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00204EB	Americium 241	AQUATIC	0	pCi/g	U
SWC001	FI	BI00204EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00204EB	Plutonium 239/240	AQUATIC	0.002	pCi/g	J
SWC001	FI	BI00204EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00204EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00205EB	Americium 241	AQUATIC	0.001	pCi/g	U
SWC001	FI	BI00205EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00205EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	J
SWC001	FI	BI00205EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00205EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00206EB	Americium 241	AQUATIC	0	pCi/g	U
SWC001	FI	BI00206EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00206EB	Plutonium 239/240	AQUATIC	0.002	pCi/g	J
SWC001	FI	BI00206EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00206EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00210EB	Americium 241	AQUATIC	0.005	pCi/g	U
SWC001	FI	BI00210EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
SWC001	FI	BI00210EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
SWC001	FI	BI00210EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00210EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00213EB	Americium 241	AQUATIC	-0.001	pCi/g	U
SWC001	FI	BI00213EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00213EB	Plutonium 239/240	AQUATIC	0.008	pCi/g	J
SWC001	FI	BI00213EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00213EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00215EB	Americium 241	AQUATIC	0.001	pCi/g	J
SWC001	FI	BI00215EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00215EB	Plutonium 239/240	AQUATIC	0.003	pCi/g	J
SWC001	FI	BI00215EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00215EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00216EB	Americium 241	AQUATIC	0	pCi/g	U
SWC001	FI	BI00216EB	Americium 241	AQUATIC	0	pCi/g	U
SWC001	FI	BI00216EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00216EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00216EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	J
SWC001	FI	BI00216EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	J
SWC001	FI	BI00216EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00216EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00216EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00216EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00217EB	Americium 241	AQUATIC	0.001	pCi/g	U
SWC001	FI	BI00217EB	Americium 241	AQUATIC	0	pCi/g	U
SWC001	FI	BI00217EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00217EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00217EB	Plutonium 239/240	AQUATIC	0.002	pCi/g	J

## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SWC001	FI	BI00217EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	J
SWC001	FI	BI00217EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00217EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00217EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00217EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00218EB	Americium 241	AQUATIC	0	pCi/g	U
SWC001	FI	BI00218EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00218EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	J
SWC001	FI	BI00218EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00218EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00219EB	Americium 241	AQUATIC	0	pCi/g	U
SWC001	FI	BI00219EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00219EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	J
SWC001	FI	BI00219EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00219EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00220EB	Americium 241	AQUATIC	0.001	pCi/g	U
SWC001	FI	BI00220EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
SWC001	FI	BI00220EB	Plutonium 239/240	AQUATIC	0.002	pCi/g	U
SWC001	FI	BI00220EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00220EB	Total Uranium	AQUATIC		pCi/g	
SWC001	FI	BI00221EB	Americium 241	AQUATIC	0	pCi/g	U
SWC001	FI	BI00221EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC001	FI	BI00221EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SWC001	FI	BI00221EB	Radium 226	AQUATIC		pCi/g	
SWC001	FI	BI00221EB	Total Uranium	AQUATIC		pCi/g	
SWC002	AV	BI00525EB	Americium 241	AQUATIC	0	pCi/g	U
SWC002	AV	BI00525EB	Americium 241	AQUATIC	0	pCi/g	UX
SWC002	AV	BI00525EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC002	AV	BI00525EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC002	AV	BI00525EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	BJ
SWC002	AV	BI00525EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
SWC002	AV	BI00525EB	Radium 226	AQUATIC		pCi/g	
SWC002	AV	BI00525EB	Radium 226	AQUATIC		pCi/g	
SWC002	AV	BI00525EB	Total Uranium	AQUATIC		pCi/g	
SWC002	AV	BI00525EB	Total Uranium	AQUATIC		pCi/g	
SWC002	FI	BI00472EB	Americium 241	AQUATIC	0.017	pCi/g	J
SWC002	FI	BI00472EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC002	FI	BI00472EB	Plutonium 239/240	AQUATIC	0.11	pCi/g	
SWC002	FI	BI00472EB	Radium 226	AQUATIC		pCi/g	
SWC002	FI	BI00472EB	Total Uranium	AQUATIC	0.36	pCi/g	
SWC002	FI	BI00473EB	Americium 241	AQUATIC	0.017	pCi/g	U
SWC002	FI	BI00473EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
SWC002	FI	BI00473EB	Plutonium 239/240	AQUATIC	0.11	pCi/g	
SWC002	FI	BI00473EB	Radium 226	AQUATIC		pCi/g	
SWC002	FI	BI00473EB	Total Uranium	AQUATIC	0.39	pCi/g	
SWC002	FI	BI00474EB	Americium 241	AQUATIC	0.009	pCi/g	J
SWC002	FI	BI00474EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
SWC002	FI	BI00474EB	Plutonium 239/240	AQUATIC	0.092	pCi/g	
SWC002	FI	BI00474EB	Radium 226	AQUATIC		pCi/g	
SWC002	FI	BI00474EB	Total Uranium	AQUATIC	0.39	pCi/g	
SWC002	HE	BI00233EB	Americium 241	AQUATIC	0.008	pCi/g	J
SWC002	HE	BI00233EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
SWC002	HE	BI00233EB	Plutonium 239/240	AQUATIC	0.037	pCi/g	B
SWC002	HE	BI00233EB	Radium 226	AQUATIC		pCi/g	
SWC002	HE	BI00233EB	Total Uranium	AQUATIC	0.17	pCi/g	J

## RAD RAW DATA LISTING 20-SEP-92

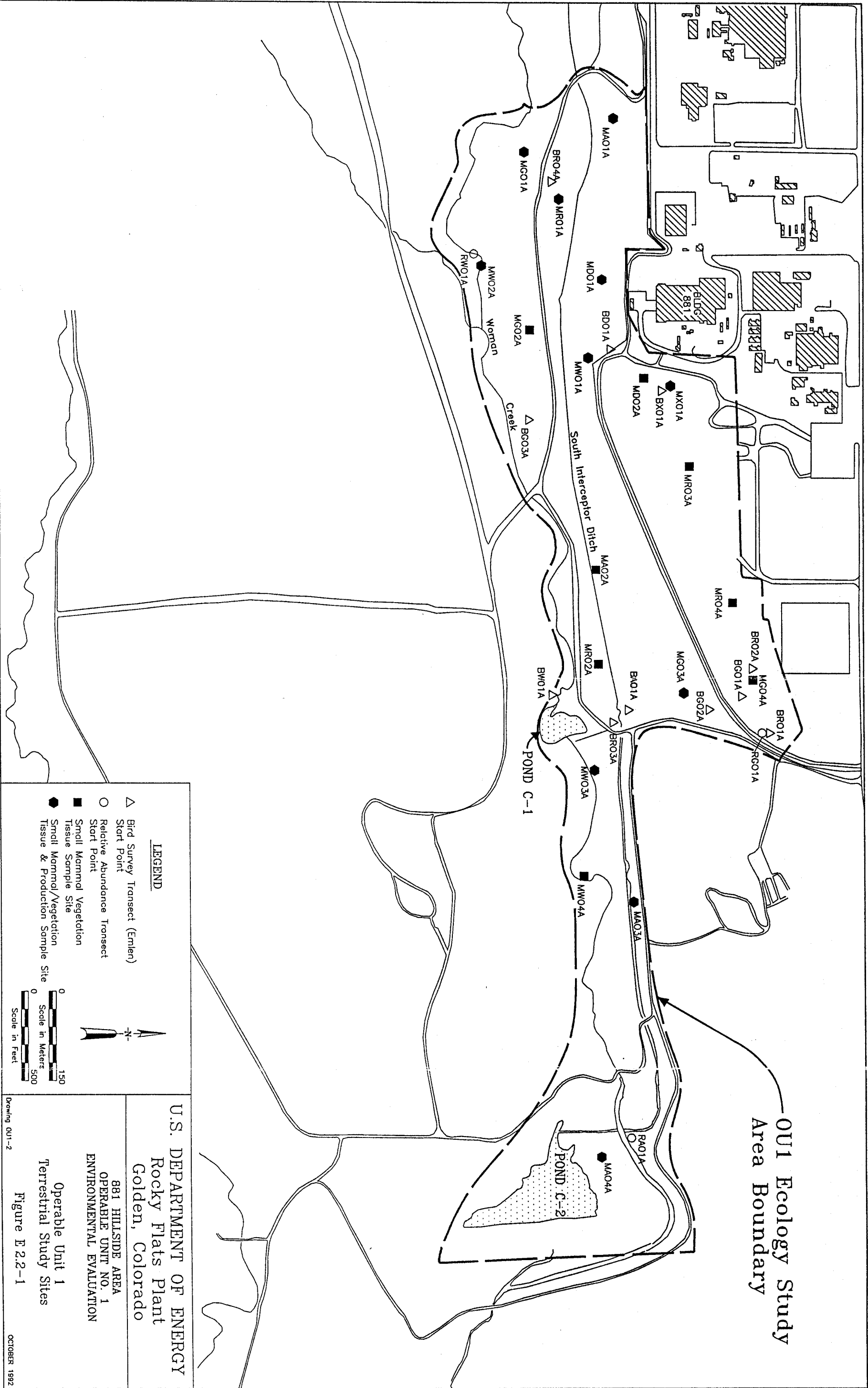
LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
SWC002	HE	BI00234EB	Americium 241	AQUATIC	0.007	pCi/g	U
SWC002	HE	BI00234EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
SWC002	HE	BI00234EB	Plutonium 239/240	AQUATIC	0.047	pCi/g	
SWC002	HE	BI00234EB	Radium 226	AQUATIC		pCi/g	
SWC002	HE	BI00234EB	Total Uranium	AQUATIC	0.18	pCi/g	J
SWC002	HE	BI00235EB	Americium 241	AQUATIC	0.009	pCi/g	J
SWC002	HE	BI00235EB	Plutonium 238	AQUATIC	0	pCi/g	U
SWC002	HE	BI00235EB	Plutonium 239/240	AQUATIC	0.037	pCi/g	B
SWC002	HE	BI00235EB	Radium 226	AQUATIC		pCi/g	
SWC002	HE	BI00235EB	Total Uranium	AQUATIC	0.12	pCi/g	J
SWC002	HE	BI00237EB	Americium 241	AQUATIC	-0.001	pCi/g	U
SWC002	HE	BI00237EB	Plutonium 238	AQUATIC	0.002	pCi/g	U
SWC002	HE	BI00237EB	Plutonium 239/240	AQUATIC	0.053	pCi/g	
SWC002	HE	BI00237EB	Radium 226	AQUATIC		pCi/g	
SWC002	HE	BI00237EB	Total Uranium	AQUATIC	0.10	pCi/g	J
WOPO02	BM	BI00496EB	Americium 241	AQUATIC	-0.002	pCi/g	U
WOPO02	BM	BI00496EB	Plutonium 238	AQUATIC	0	pCi/g	U
WOPO02	BM	BI00496EB	Plutonium 239/240	AQUATIC	0.021	pCi/g	
WOPO02	BM	BI00496EB	Radium 226	AQUATIC		pCi/g	
WOPO02	BM	BI00496EB	Total Uranium	AQUATIC	0.20	pCi/g	
WOPO02	BM	BI00497EB	Americium 241	AQUATIC	0.004	pCi/g	U
WOPO02	BM	BI00497EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
WOPO02	BM	BI00497EB	Plutonium 239/240	AQUATIC	0.025	pCi/g	
WOPO02	BM	BI00497EB	Radium 226	AQUATIC		pCi/g	
WOPO02	BM	BI00497EB	Total Uranium	AQUATIC		pCi/g	
WOPO02	FI	BI00480EB	Americium 241	AQUATIC	0.001	pCi/g	U
WOPO02	FI	BI00480EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
WOPO02	FI	BI00480EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
WOPO02	FI	BI00480EB	Radium 226	AQUATIC		pCi/g	
WOPO02	FI	BI00480EB	Total Uranium	AQUATIC	0.15	pCi/g	J
WOPO02	FI	BI00481EB	Americium 241	AQUATIC	-0.001	pCi/g	U
WOPO02	FI	BI00481EB	Plutonium 238	AQUATIC	0	pCi/g	U
WOPO02	FI	BI00481EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
WOPO02	FI	BI00481EB	Radium 226	AQUATIC		pCi/g	
WOPO02	FI	BI00481EB	Total Uranium	AQUATIC	0.29	pCi/g	
WOPO02	FI	BI00482EB	Americium 241	AQUATIC	0.007	pCi/g	J
WOPO02	FI	BI00482EB	Plutonium 238	AQUATIC	0	pCi/g	U
WOPO02	FI	BI00482EB	Plutonium 239/240	AQUATIC	0	pCi/g	U
WOPO02	FI	BI00482EB	Radium 226	AQUATIC		pCi/g	
WOPO02	FI	BI00482EB	Total Uranium	AQUATIC	0.38	pCi/g	
WOPO02	FI	BI00483EB	Americium 241	AQUATIC	0.003	pCi/g	U
WOPO02	FI	BI00483EB	Americium 241	AQUATIC	-0.001	pCi/g	U
WOPO02	FI	BI00483EB	Plutonium 238	AQUATIC	0.002	pCi/g	J
WOPO02	FI	BI00483EB	Plutonium 238	AQUATIC	0	pCi/g	U
WOPO02	FI	BI00483EB	Plutonium 239/240	AQUATIC	0.003	pCi/g	J
WOPO02	FI	BI00483EB	Plutonium 239/240	AQUATIC	-0.001	pCi/g	U
WOPO02	FI	BI00483EB	Radium 226	AQUATIC		pCi/g	
WOPO02	FI	BI00483EB	Radium 226	AQUATIC		pCi/g	
WOPO02	FI	BI00483EB	Total Uranium	AQUATIC	0.31	pCi/g	
WOPO02	FI	BI00483EB	Total Uranium	AQUATIC	0.21	pCi/g	
WORI01	BM	BI00487EB	Americium 241	AQUATIC	0	pCi/g	U
WORI01	BM	BI00487EB	Plutonium 238	AQUATIC	0	pCi/g	U
WORI01	BM	BI00487EB	Plutonium 239/240	AQUATIC	0.024	pCi/g	
WORI01	BM	BI00487EB	Radium 226	AQUATIC		pCi/g	
WORI01	BM	BI00487EB	Total Uranium	AQUATIC	0.093	pCi/g	J

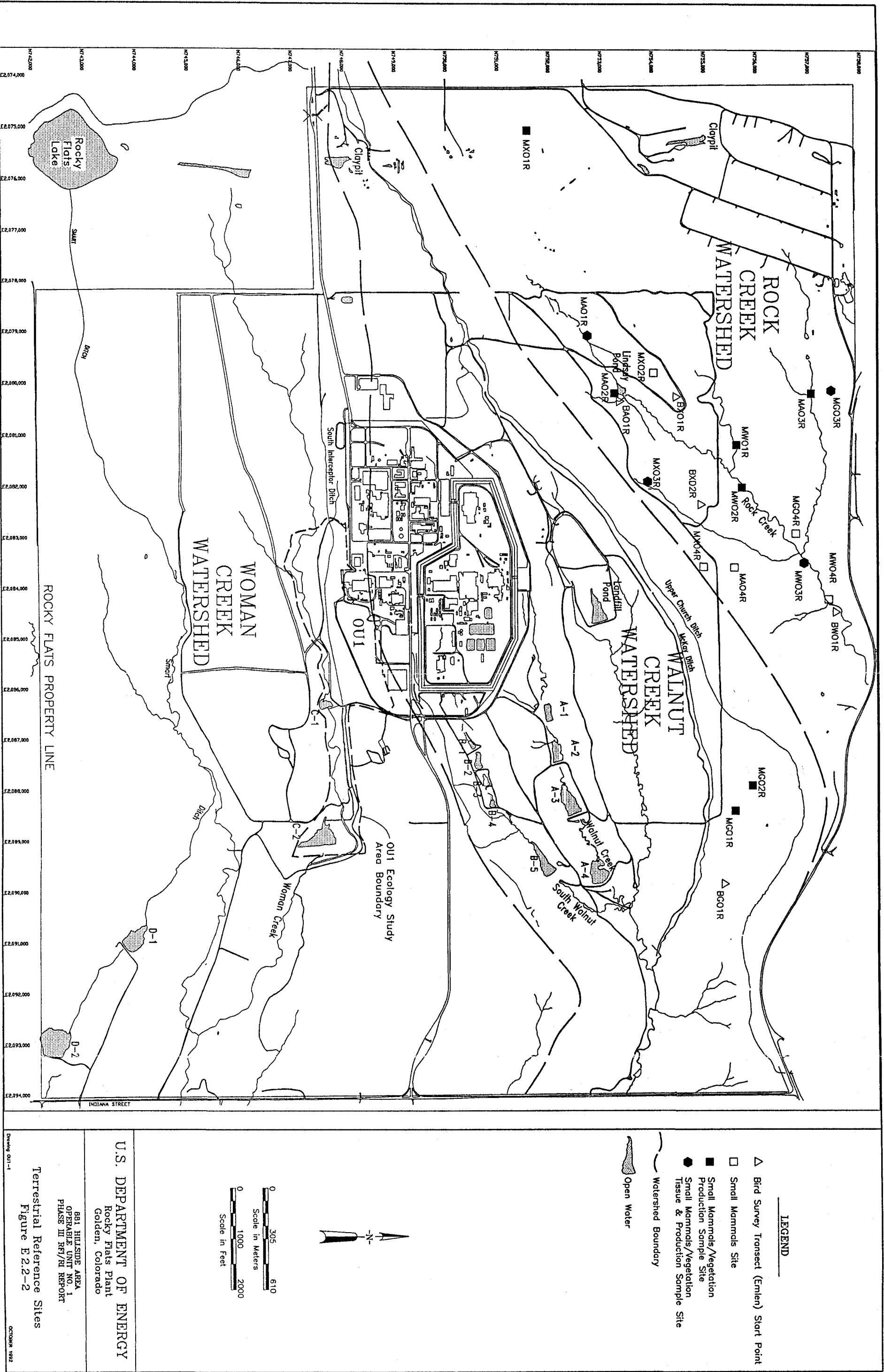


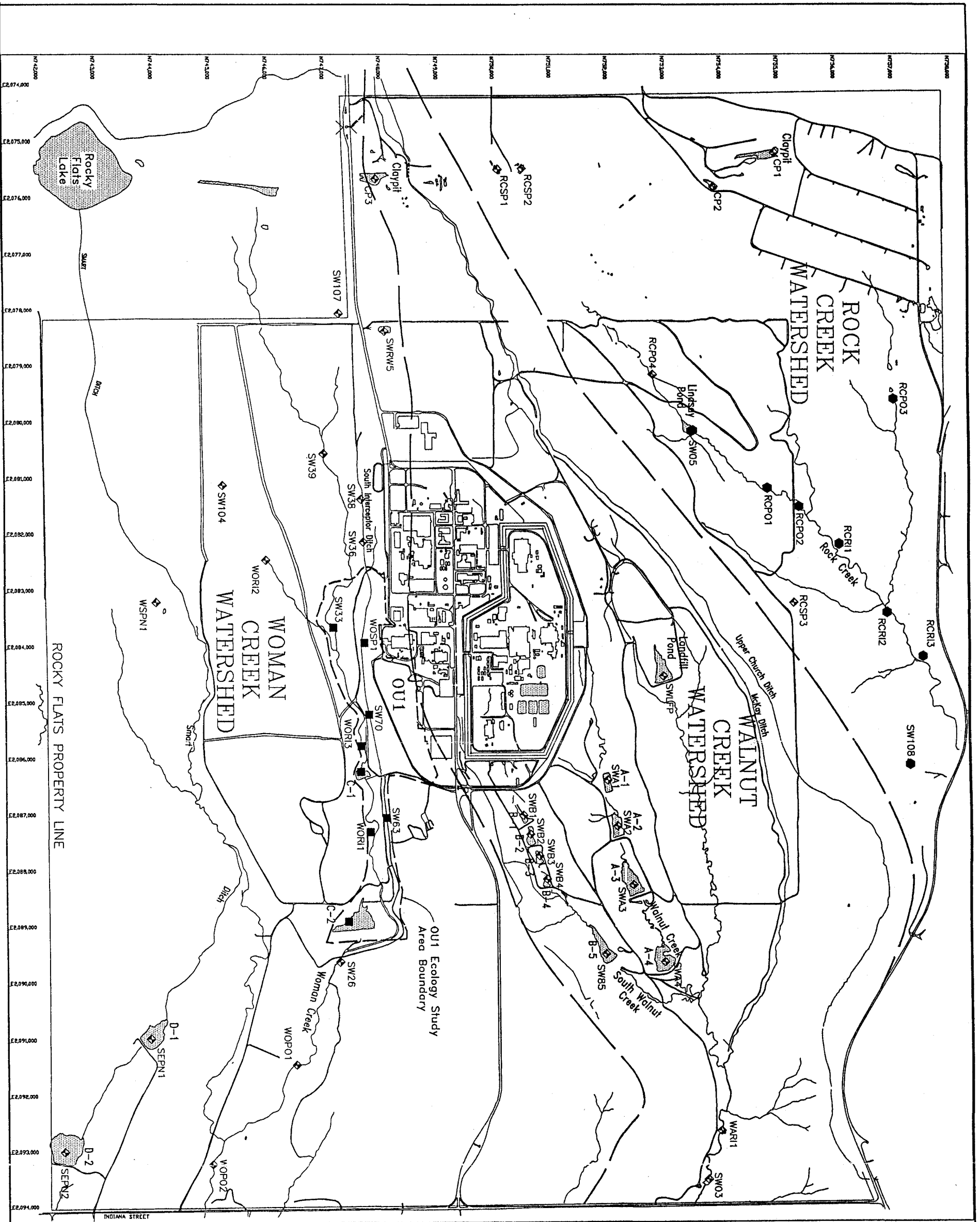
## RAD RAW DATA LISTING 20-SEP-92

LOCATION	SAM	PRJSMPNO	ANALYTE	COMMTYP	RESULT	UNITS	QUAL
WORI01	FI	BI00488EB	Americium 241	AQUATIC	0.002	pCi/g	U
WORI01	FI	BI00488EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
WORI01	FI	BI00488EB	Plutonium 239/240	AQUATIC	0.034	pCi/g	
WORI01	FI	BI00488EB	Radium 226	AQUATIC		pCi/g	
WORI01	FI	BI00488EB	Total Uranium	AQUATIC	0.20	pCi/g	
WORI01	FI	BI00489EB	Americium 241	AQUATIC	-0.001	pCi/g	U
WORI01	FI	BI00489EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
WORI01	FI	BI00489EB	Plutonium 239/240	AQUATIC	0.004	pCi/g	J
WORI01	FI	BI00489EB	Radium 226	AQUATIC		pCi/g	
WORI01	FI	BI00489EB	Total Uranium	AQUATIC	0.19	pCi/g	J
WORI01	FI	BI00490EB	Americium 241	AQUATIC	-0.003	pCi/g	U
WORI01	FI	BI00490EB	Plutonium 238	AQUATIC	0	pCi/g	U
WORI01	FI	BI00490EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
WORI01	FI	BI00490EB	Radium 226	AQUATIC		pCi/g	
WORI01	FI	BI00490EB	Total Uranium	AQUATIC	0.086	pCi/g	J
WORI01	FI	BI00491EB	Americium 241	AQUATIC	0	pCi/g	U
WORI01	FI	BI00491EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
WORI01	FI	BI00491EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
WORI01	FI	BI00491EB	Radium 226	AQUATIC		pCi/g	
WORI01	FI	BI00491EB	Total Uranium	AQUATIC	0.094	pCi/g	J
WORI01	FI	BI00492EB	Americium 241	AQUATIC	0.007	pCi/g	U
WORI01	FI	BI00492EB	Plutonium 238	AQUATIC	0	pCi/g	U
WORI01	FI	BI00492EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
WORI01	FI	BI00492EB	Radium 226	AQUATIC		pCi/g	
WORI01	FI	BI00492EB	Total Uranium	AQUATIC	0.10	pCi/g	J
WORI01	FI	BI00493EB	Americium 241	AQUATIC	0.003	pCi/g	U
WORI01	FI	BI00493EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
WORI01	FI	BI00493EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
WORI01	FI	BI00493EB	Radium 226	AQUATIC		pCi/g	
WORI01	FI	BI00493EB	Total Uranium	AQUATIC	0.052	pCi/g	J
WORI01	FI	BI00494EB	Americium 241	AQUATIC	-0.003	pCi/g	U
WORI01	FI	BI00494EB	Plutonium 238	AQUATIC	0.001	pCi/g	U
WORI01	FI	BI00494EB	Plutonium 239/240	AQUATIC	0.003	pCi/g	U
WORI01	FI	BI00494EB	Radium 226	AQUATIC		pCi/g	
WORI01	FI	BI00494EB	Total Uranium	AQUATIC	0.15	pCi/g	J
WORI03	FI	BI00495EB	Americium 241	AQUATIC	-0.003	pCi/g	U
WORI03	FI	BI00495EB	Plutonium 238	AQUATIC	0	pCi/g	U
WORI03	FI	BI00495EB	Plutonium 239/240	AQUATIC	0.001	pCi/g	U
WORI03	FI	BI00495EB	Radium 226	AQUATIC		pCi/g	
WORI03	FI	BI00495EB	Total Uranium	AQUATIC	0.057	pCi/g	J
WORI03	FI	BI00498EB	Americium 241	AQUATIC	0	pCi/g	U
WORI03	FI	BI00498EB	Plutonium 238	AQUATIC	-0.001	pCi/g	U
WORI03	FI	BI00498EB	Plutonium 239/240	AQUATIC	0.002	pCi/g	J
WORI03	FI	BI00498EB	Radium 226	AQUATIC		pCi/g	
WORI03	FI	BI00498EB	Total Uranium	AQUATIC	0.094	pCi/g	J
WORI03	FI	BI00499EB	Americium 241	AQUATIC	-0.002	pCi/g	U
WORI03	FI	BI00499EB	Plutonium 238	AQUATIC	-0.002	pCi/g	U
WORI03	FI	BI00499EB	Plutonium 239/240	AQUATIC	0.002	pCi/g	U
WORI03	FI	BI00499EB	Radium 226	AQUATIC		pCi/g	
WORI03	FI	BI00499EB	Total Uranium	AQUATIC	0.17	pCi/g	J

765 rows selected.

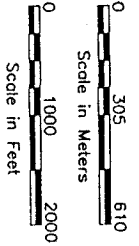
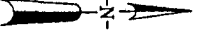






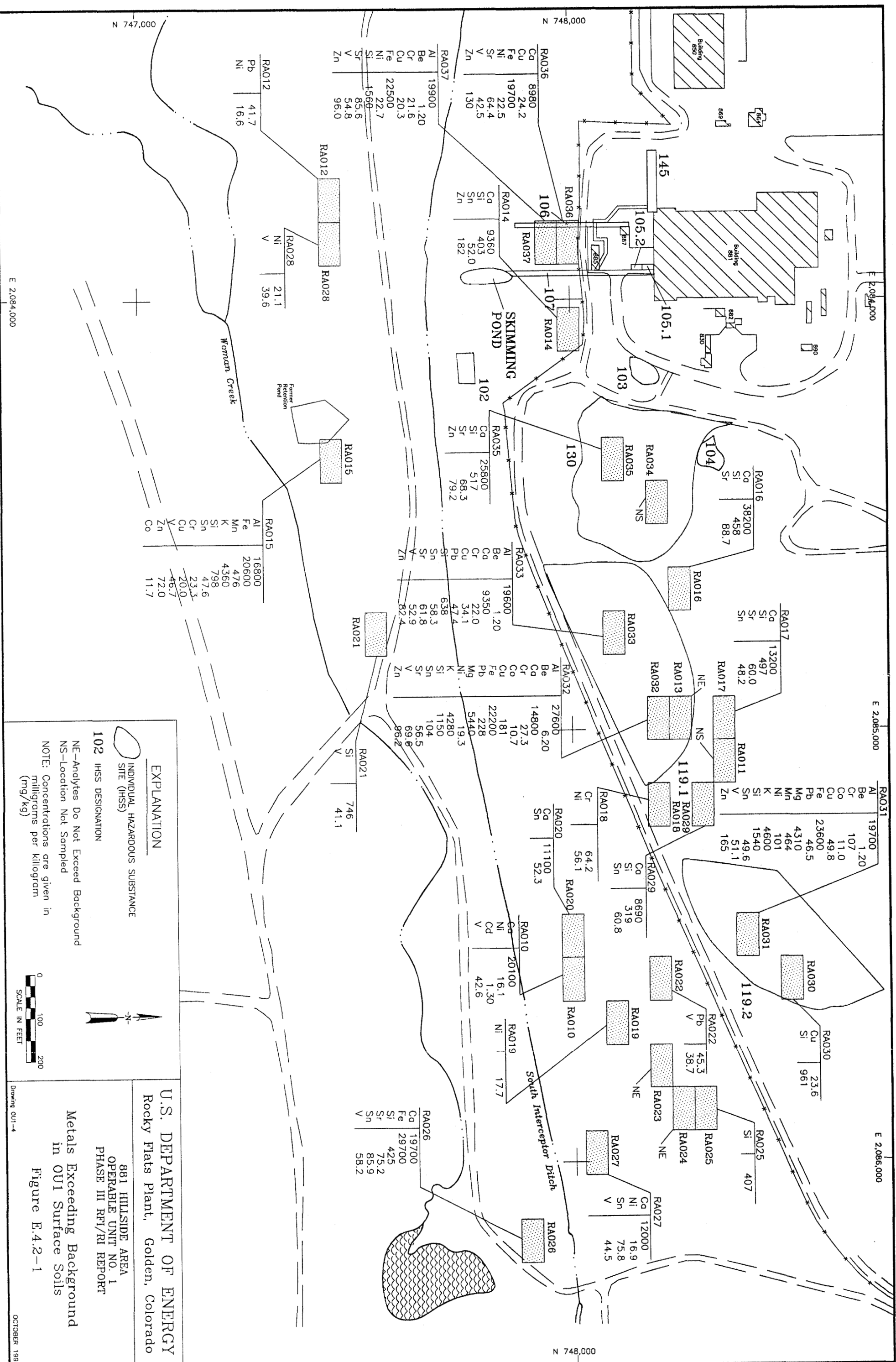
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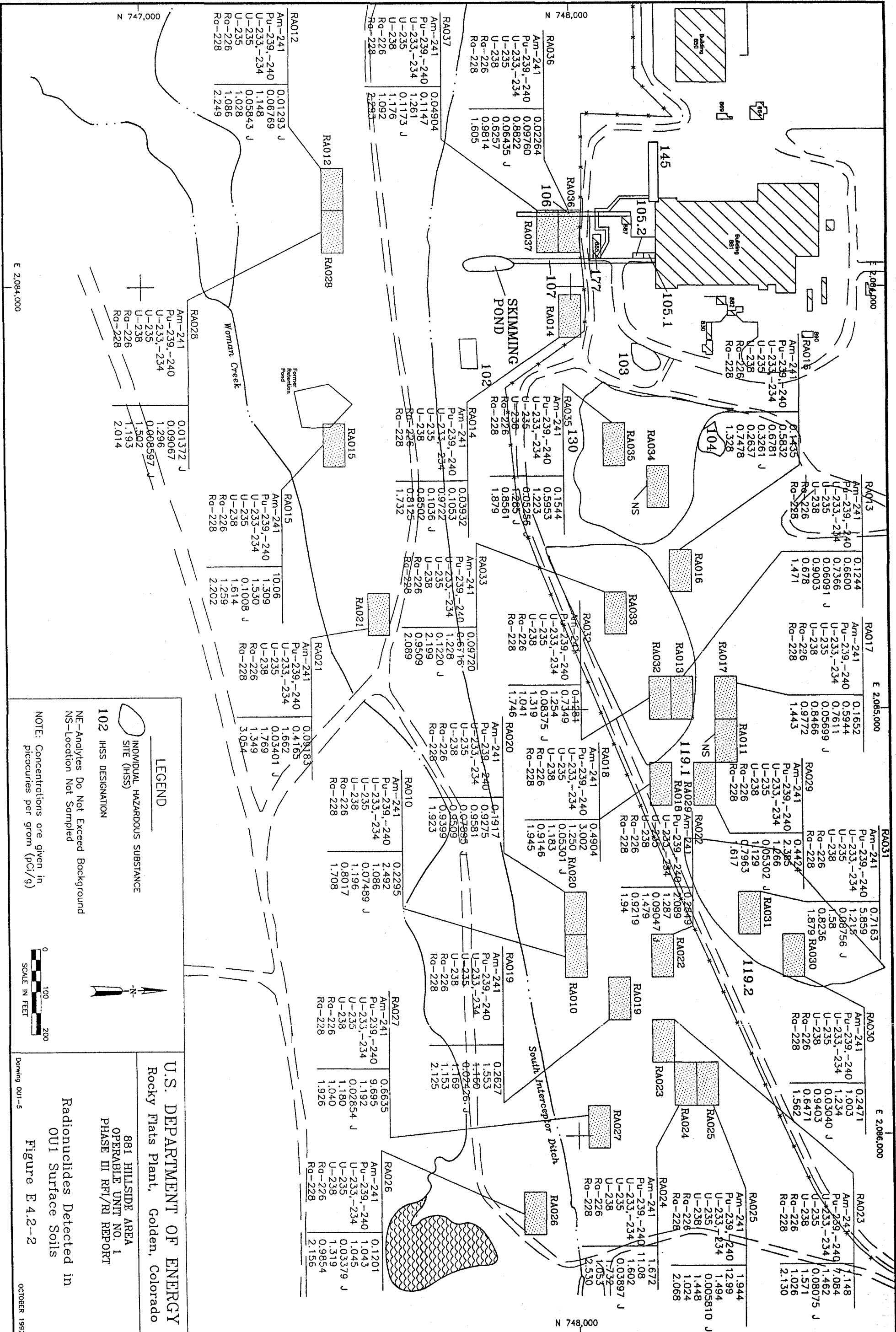
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- OUI Aquatic Reference Sites
- ◆ Other Sample Sites
- Watershed Boundary
- Open Water

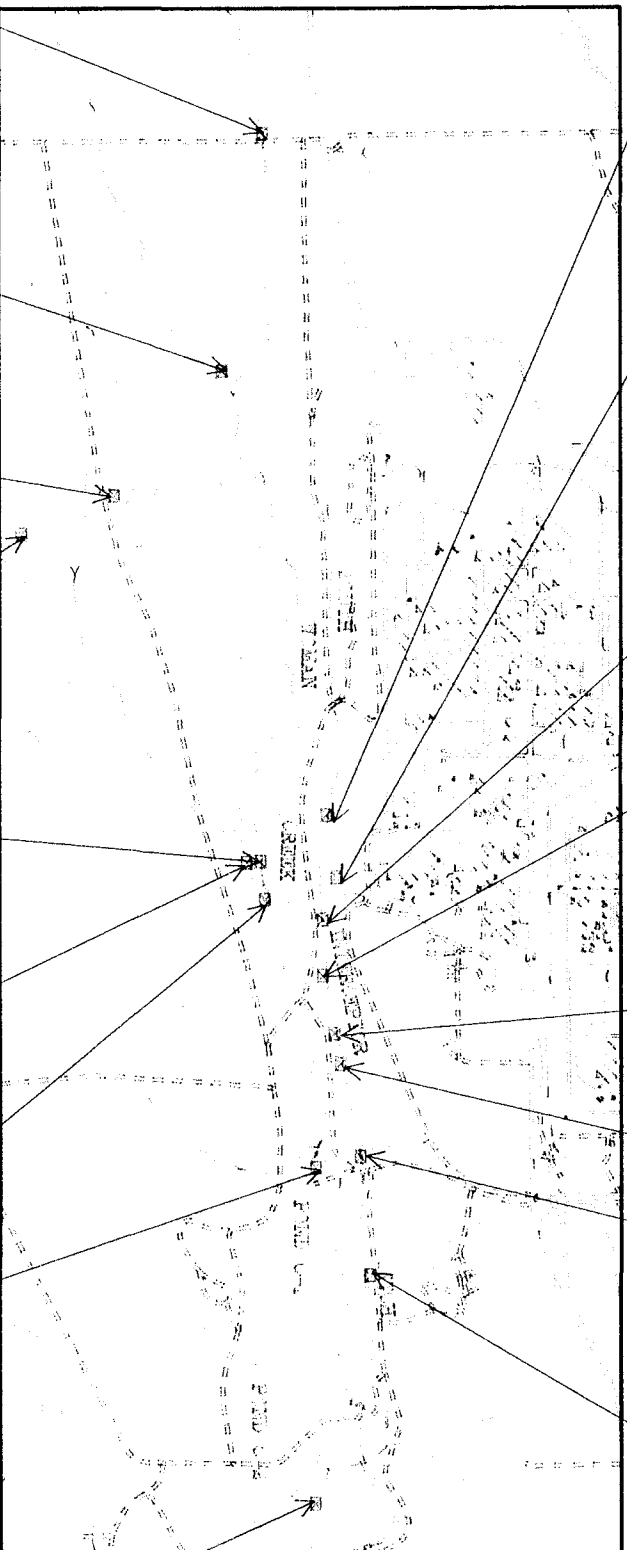


U.S. DEPARTMENT OF ENERGY  
Rocky Flats Plant  
Golden, Colorado

881 HILLSIDE AREA  
OPERABLE UNIT NO. 1  
PHASE III RFI/RI REPORT  
Aquatic Study Sites and Aquatic  
Reference Sites  
Figure E.2.2-3  
October 1992







SW046			
ANALYTE	MAX CONC	RFP BCKGRND	
Cr	11.4	10	
U	5.24	2.0	

SW035			
ANALYTE	MAX CONC	RFP BCKGRND	
Cr	14.3	10	
Cu	28.1	18	
Hg	1.0	0.3	

SW031			
ANALYTE	MAX CONC	RFP BCKGRND	
Cr	18	10	
Hg	0.56	0.3	
U	2.31	2.0	

SW067	
ANALYTE	
NONE	

SW069			
ANALYTE	MAX CONC	RFP BCKGRND	
Al	510	295	
Cr	13	10	
U	2.69	2.0	

SW070			
ANALYTE	MAX CONC	RFP BCKGRND	
Al	450	295	
Cr	63	10	
U	3.67	2.0	

SW064			
ANALYTE	MAX CONC	RFP BCKGRND	
Al	2400	295	
Pb	38.4	4	
Fe	2300	113	

SW030	
ANALYTE	
NONE	

SW107			
ANALYTE	MAX CONC	RFP BCKGRND	
Hg	0.39	0.3	

SW041			
ANALYTE	MAX CONC	RFP BCKGRND	
Cu	28	18	

SW080			
ANALYTE	MAX CONC	RFP BCKGRND	
Fe	16400	113	
Mn	376	292	
Hg	0.32	0.3	

SW104	
ANALYTE	
NONE	

SW033			
ANALYTE	MAX CONC	RFP BCKGRND	
Cu	20.1	18	
Pb	4.5	4	

SW034			
ANALYTE	MAX CONC	RFP BCKGRND	
Cr	12.9	13.2	
Pb	13.2	4	

SW032	
ANALYTE	
NONE	

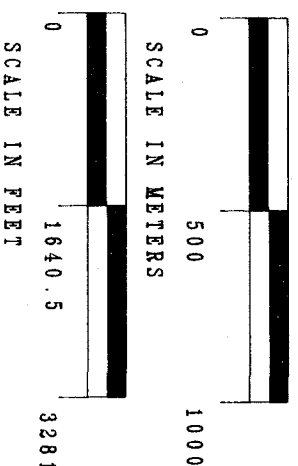
C-1	
ANALYTE	
NONE	

C-2			
ANALYTE	MAX CONC	RFP BCKGRND	
Al	770	295	
As	5.2	3	
Pb	6.2	4	
Mn	730	292	

## MAP LEGEND

- SURFACE WATER STATIONS
- STREAMS, DITCHES, DRAINAGE FEATURES
- PAVED ROADS
- DIRT ROADS
- BUILDINGS

Data presented are the COCs for which surface water concentration exceeded RFP background, and the maximum concentration detected (micrograms per liter for metals; picocuries per liter for radionuclides). Data from January 1990 to December 1991.



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Rocky Flats Plant, Golden, Colorado

SUMMARY OF  
CONTAMINANTS OF CONCERN  
IN SURFACE WATER AT 001:  
DISSOLVED METALS AND  
RADIONUCLIDES

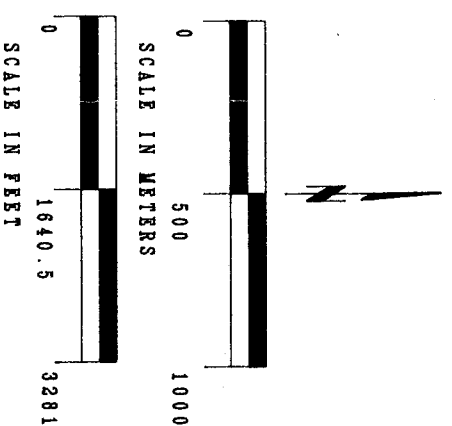
FIGURE E.4.2-3  
DRAWING 001-6  
OCTOBER 1992



# MAP LEGEND

- SURFACE WATER STATIONS
- STREAMS, DITCHES, DRAINAGE FEATURES
- PAVED ROADS
- DIRT ROADS
- BUILDINGS

Data presented are the COCs for which surface water concentration exceeded RFP background and Colorado water quality standards, the number of monthly samples which exceeded chronic and acute standards, and the maximum concentration detected (micrograms per liter). Data from January 1990 to December 1991.

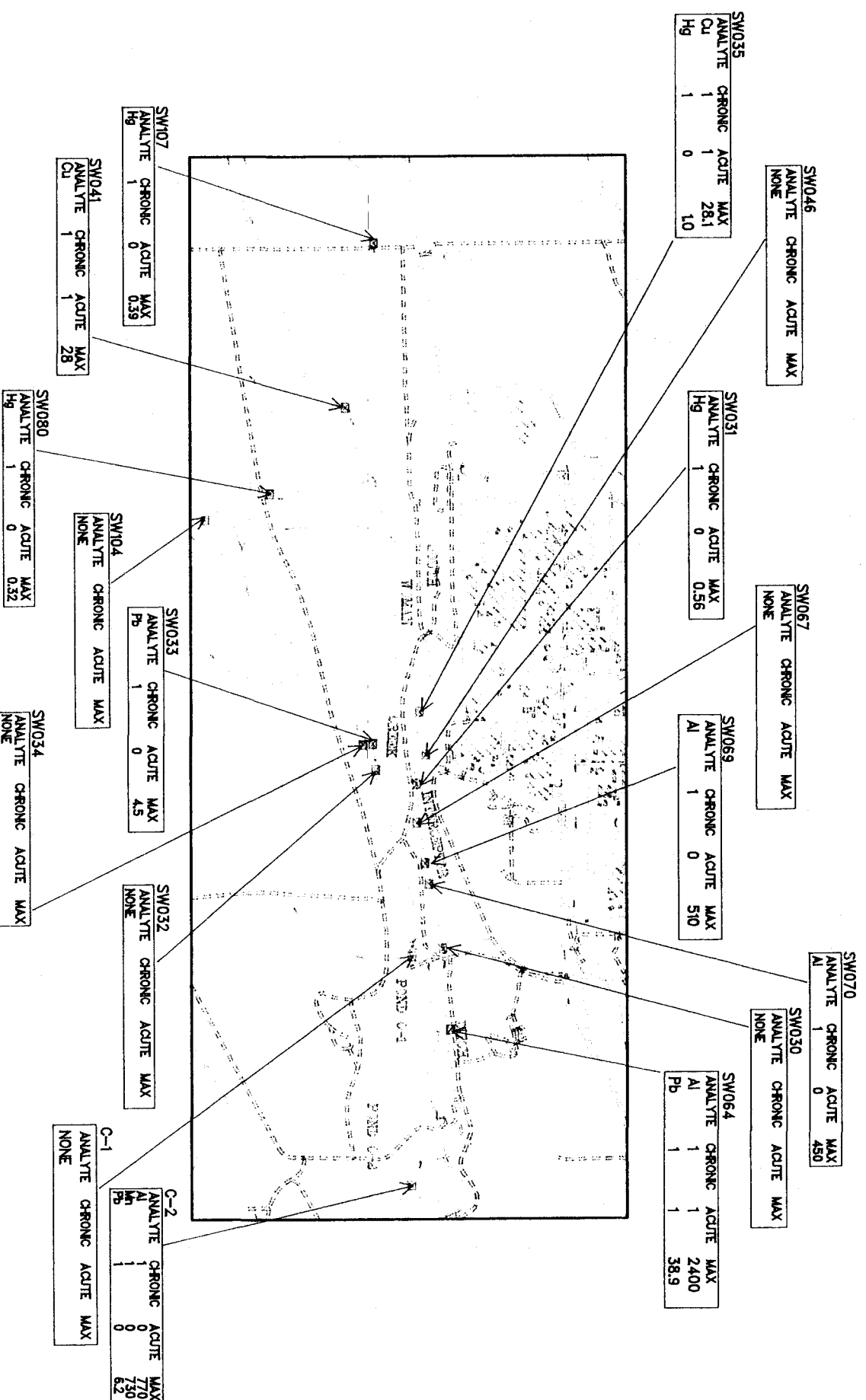


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SUMMARY OF  
CONTAMINANTS OF CONCERN  
IN SURFACE WATER AT 0U1:  
EXCEEDENCE OF WATER  
QUALITY STANDARDS

FIGURE E.4.2-4

DRAWING 0U1-7 OCTOBER 1992





SW033	
Ceriodaphnia	Fathead Minnows
No. Survivors	18
pH Range	7.3-7.8
Alkalinity	138
Hardness	124

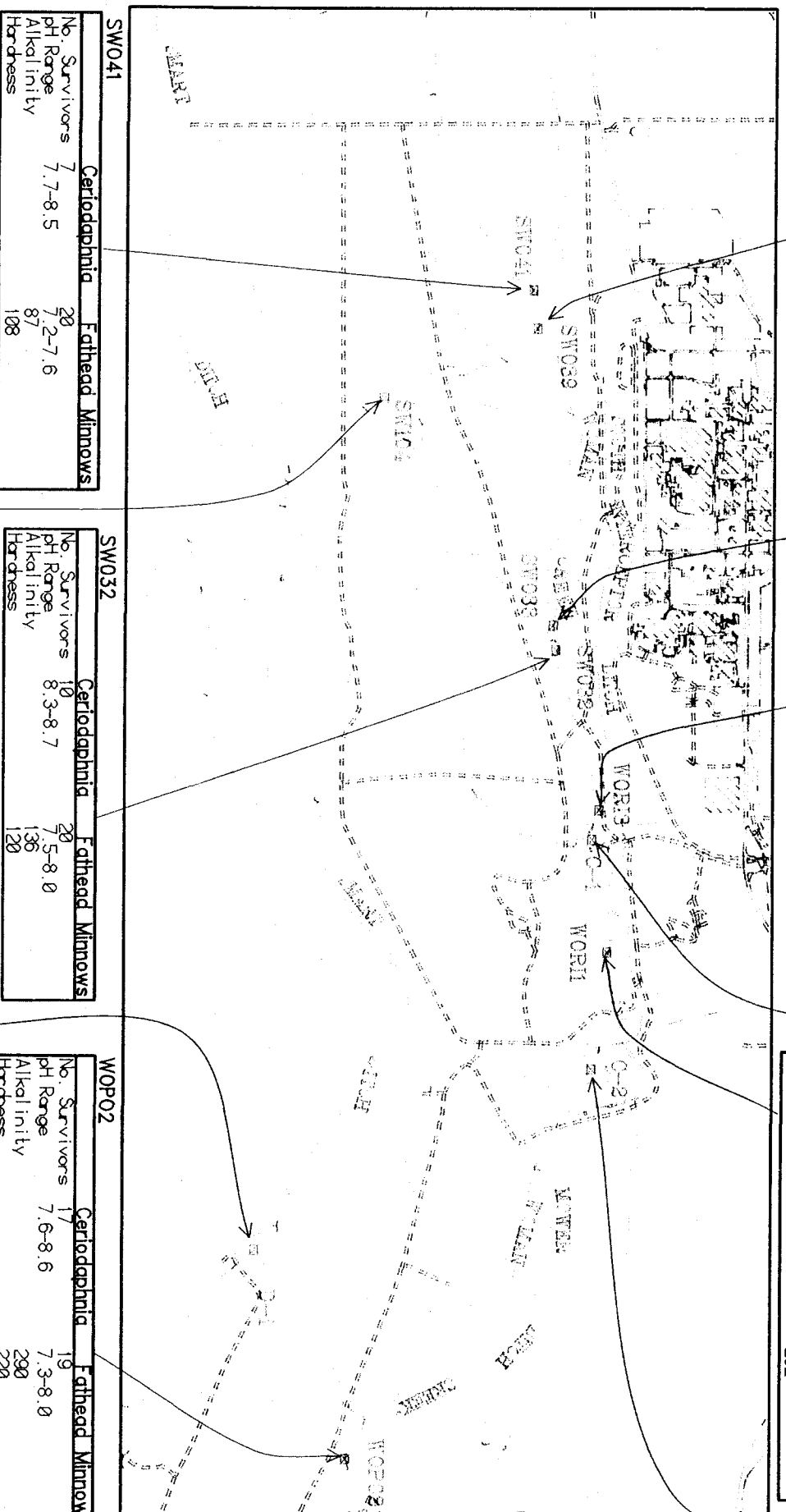
C-1	
Ceriodaphnia	Fathead Minnows
No. Survivors	20
pH Range	7.6-8.5
Alkalinity	165
Hardness	170
Total Ammonia	ND

C-2	
Ceriodaphnia	Fathead Minnows
No. Survivors	19
pH Range	7.7-8.6
Alkalinity	191
Hardness	173
Total Ammonia	ND

SW039	
Ceriodaphnia	Fathead Minnows
No. Survivors	18
pH Range	7.2-7.6
Alkalinity	90
Hardness	86

WOR3	
Ceriodaphnia	Fathead Minnows
No. Survivors	11
pH Range	8.1-8.6
Alkalinity	138
Hardness	150

WOR1	
Ceriodaphnia	Fathead Minnows
No. Survivors	15
pH Range	8.0-8.6
Alkalinity	200
Hardness	202



SW104	
Ceriodaphnia	Fathead Minnows
No. Survivors	10
pH Range	7.0-7.5
Alkalinity	88
Hardness	108

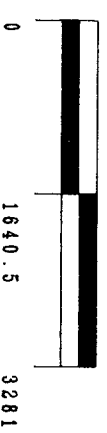
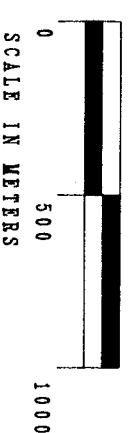
D-1	
Ceriodaphnia	Fathead Minnows
No. Survivors	20
pH Range	8.5-8.7
Alkalinity	219
Hardness	181
Total Ammonia	ND

## MAP LEGEND

- SAMPLE LOCATIONS
- PAVED ROADS
- DIRT ROADS
- ▨ BUILDINGS

Toxicity screen results reported as survivors out of 20 organisms.

UNITS: ALKALINITY (mg/l as  $\text{CaCO}_3$ )  
UNITS: HARDNESS (mg/l as  $\text{CaCO}_3$ )



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RESULTS OF AQUATIC  
TOXICITY SCREENING,  
WOMAN CREEK - 1991

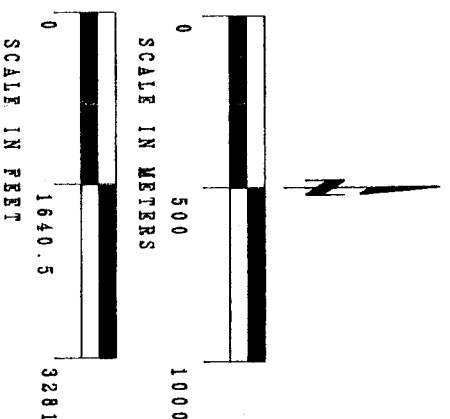
FIGURE E.4.2-5

DRAWING 001-9 OCTOBER 1992

# MAP LEGEND

- SEDIMENT STATIONS
- STREAMS, DITCHES  
DRAINAGE FEATURES
- PAVED ROADS
- DIRT ROADS
- BUILDINGS

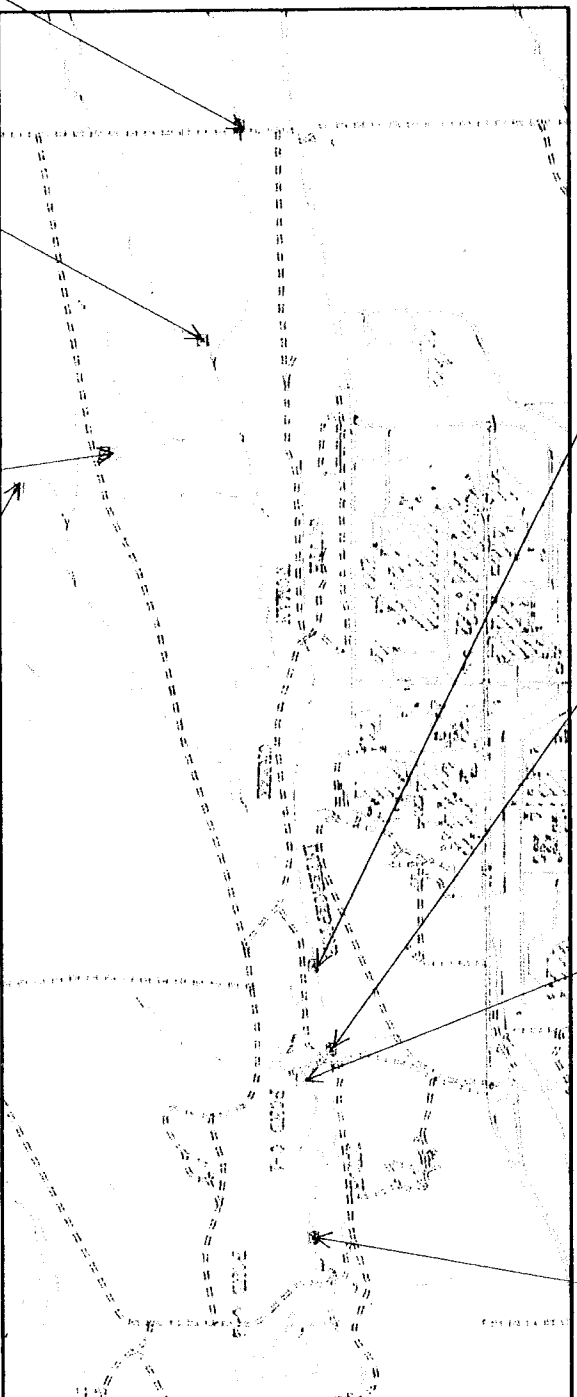
Data presented are the  
COCs for which sediment  
concentration exceeded RfP  
background, and the maximum  
concentration detected (mg/kg-  
metals; pci/g-radiionuclides).



U.S. DEPARTMENT of ENERGY  
Rocky Flats Plant, Golden, Colorado

SUMMARY OF  
CONTAMINANTS OF CONCERN  
IN SEDIMENTS AT OU1:  
METALS AND  
RADIIONUCLIDES

FIGURE E.4.2-6  
DRAWING OU1-8  
OCTOBER 1992



SED028	ANALYTE	MAX CONC	RfP BCKGRND
NONE			

SED031	ANALYTE	MAX CONC	RfP BCKGRND
NONE			

SED027	ANALYTE	MAX CONC	RfP BCKGRND
AI		45400	20840

SED026	ANALYTE	MAX CONC	RfP BCKGRND
NONE			

SED006	ANALYTE	MAX CONC	RfP BCKGRND
NONE			

SED007	ANALYTE	MAX CONC	RfP BCKGRND
NONE			

SED008	ANALYTE	MAX CONC	RfP BCKGRND
As		30.9	11.5
Pb		65300	33548

SED009	ANALYTE	MAX CONC	RfP BCKGRND
As		26.4	11.5
Pb		65300	33548